Risk Assessment for Contaminated Sites in Europe

CARACAS

Concerted Action on Risk Assessment for Contaminated Sites in the European Union 1996–1998

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Risk Assessment for Contaminated Sites in Europe

Volume 1 Scientific Basis

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Contents

Foreword

The pollution of soil and groundwater caused by abandoned waste disposal sites and contaminated industrial areas is a complex environmental problem in all industrialised countries. Within the European Union, several Member States have developed or are currently developing frameworks and procedures for assessing and managing the risks posed by contaminated sites. In order to coordinate and concentrate these efforts and to support scientific cooperation between European countries, CARACAS, the Concerted Action on Risk Assessment for Contaminated Sites, was initiated.

CARACAS, which started in February 1996, is funded by the EC Environment and Climate R&D Programme in order to tackle the problem of contaminated land. It was initiated by the German Federal Ministry for the Environment and coordinated by the Federal Environmental Agency. It brings together the combined knowledge of academics and government experts from 16 European countries. The CARACAS work focuses on the coordination of current research initiatives on contaminated land risk assessment in Europe, and on the definition of scientific priorities for future R&D programmes in order to improve the scientific basis for assessing risks from contaminated sites.

During the course of CARACAS a fruitful cooperation between scientists, representatives of the participating countries and the European Commission, and various national and international initiatives has been developed. This book is one of the major outputs from this scientific partnership. It summarises the conclusions and recommendations drawn by CARACAS scientists for various research areas related to risk assessment of contaminated land. It is a unique reference for the practical state-of-the-art on risk assessment in Europe, and explains perceived research needs in the context of current approaches for contaminated land risk assessment in European countries.

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Chapter 1

General introduction

1.1 Background to this book

Past and present human activities that introduce contaminants into soil and groundwater have resulted in some 750,000 sites across Europe with suspected contamination. Some of these sites may endanger water resources, ecosystems and/or human health. Uncertainties about the nature and significance of chemical contamination can be a major stumbling block hindering sustainable development in cities and rural areas, and increasing pressures on greenfield sites.

Better methods are therefore needed for assessing the likely impacts on humans and the environment, to confirm that sites are fit for their current or intended uses, and to guide the remedial actions needed to ensure fitness for use, the conservation of water resources and a reduced burden of aftercare for future generations.

CARACAS, the *Concerted Action on Risk Assessment for Contaminated Sites in the European Union,* was established in February 1996 as part of the EC Environment and Climate RTD Programme to tackle the problem of contaminated land. It brings together the combined knowledge of academics, government representatives and other experts from all EU Member States plus Norway and Switzerland. The CARACAS Network has also established and maintained links with other international initiatives related to contaminated land, including:

- NICOLE, the industry-led Network for Industrially Contaminated Land in Europe, which was also established under the EC Environment and Climate Programme
- The European Topic Centre on Soil (ETC/S) which was established by the European Environment Agency in 1996
- RACE, the Risk Abatement Centre for Central and Eastern Europe
- ISO Technical Committee 190 (SC7: Soil Quality, Soil and Site Assessment)

- The Ad Hoc International Working Group on Contaminated Land
- The NATO/CCMS Pilot Study programme.

Together with NICOLE, CARACAS has identified and published a brief list of research needs for contaminated land assessment and remediation.[1]

This book has been prepared by CARACAS participants to review the scientific basis and explain the perceived research needs in the context of current approaches for contaminated land risk assessment in European countries. It is not a manual for risk assessment. This volume focuses on scientific aspects of risk assessment and on research needs. The policy frameworks in which risk assessment and risk management are carried out in the participating countries are described in Volume 2, which is scheduled for publication later in 1998. Based on analysis of the current situation, priority research needs have been identified and classified into two main categories:

1. the nature of contaminated land, which deals with the characterisation of soil pollution, including its impact on water resources and other parts of the environment; and
2. the relationship between soil contamination and *fitness for use*, which deals with the conditions for sustainable landuse in urban and rural areas.

1.2 Outline of chapter contents

The subject of risk assessment is introduced in Chapter 2 'Fundamental concepts of risk assessment'. After a brief discussion about risk and risk assessment, it provides a framework for the rest of the book where the various scientific aspects of risk assessment are dealt with in more detail. Chapter 2 also identifies a number of research needs at a general methodological level, especially concerning integration of the various components of risk assessment. These research gaps do not imply that using risk assessment in its present state is not worth while. On the contrary, risk assessment has already proved to be a very useful tool in contaminated land management so long as users are aware of its limitations.

Risk assessment is mostly discussed in terms of sources, pathways and receptors. Soil contamination is a hazard, which may be a source of risk if toxic substances reach receptors by various pathways. The source–pathway–receptor concept in risk assessment is used as an organising principle for this book.

Chapters 3 and 4 consider the science needed to characterise the adverse effects on receptors. Human toxicology (Chapter 3) and ecotoxicology (Chapter 4) are reviewed with special reference to their use in contaminated land risk assessment. Many of the human toxicology data used in contaminated land risk assessment were derived from animal experiments performed for different purposes, such as establishing acceptable daily intakes of additives and environmental contaminants in food. Toxicological data based on exposure to soil contamination are generally lacking. In order to use available information on human toxicity it is usually assumed that adverse effects of exposure to soil contaminants are comparable to those resulting from exposure to the same substances in food. Chapter 3 therefore discusses the bioavailability of contaminants relative to the study or studies on which the toxicity criteria were based. This is recognised as one of the most important problems in the assessment of human health risks from exposure to contaminated land.

Chapter 4 reviews current and promising future approaches in ecological risk assessment. The suggestive title 'Ecosystem health' instead of the more traditional 'ecotoxicological risk assessment' is used to encourage the integration of ecotoxicology with ecosystem theory. Until now the ecotoxicological approach has been almost entirely based on laboratory toxicity experiments. There is at present no ecosystem theory that can serve as a framework for interpretation of laboratory data. Although human health risk assessment is also largely based on laboratory experiments with animals, there is a framework for interpretation in medicine, sociology and psychology which is lacking in the ecological approach. Compared with human health risk assessment, ecological risk assessment is a relatively new field of interest. Whereas the state of the art in human health risk assessment can be fairly readily based on a synthesis of scientific reviews, this is much more difficult for ecological risk assessment in the context of contaminated land. Moreover, human health risks only concern one species, whereas ecological risk has to address the health of ecosystems with a multitude of species, populations and communities. It was therefore decided to include in Chapter 4 a fairly detailed review of the primary literature. Further development of bioassays to supplement chemical analysis, and the validation of extrapolations from single-species toxicity testing in the laboratory to real-world situations in the field, are identified as important subjects to be addressed in future research programmes.

Characterisation of the source, usually contaminated soil or groundwater, is discussed in Chapter 5. This chapter addresses issues such as investigation planning, sampling, chemical analysis and quality control,

and compares the various approaches used in the participating countries. Site investigations should provide the necessary and sufficient data for exposure analysis and risk assessment, and must also quantify the uncertainties associated with site characterisation. The linking of site investigation to exposure analysis and the evaluation of uncertainties needs further development in most countries, and would benefit from interdisciplinary research.

The exposure pathways by which soil contamination may lead to adverse effects are addressed in Chapter 6, on the transport and fate of contaminants. In the assessment of ecosystem health risk the distinction between exposure and effect assessment is not always made. Issues concerning exposure of, or in, ecosystems are addressed in Chapter 3. The focus in Chapter 6 is on the principles needed to understand the dispersion and attenuation of pollutants in groundwater. In view of the general European policy of protecting groundwater resources as implied by the EU Groundwater Directive, this is also important for preventing soil pollution. Much less is known about the transport and fate of contamination in the unsaturated zone of the soil. Understanding of these processes is, however, of the utmost importance for contaminated land risk assessment.

Chapter 7 discusses the use of *models* in contaminated land risk assessment. Models are mostly used to predict exposure on the basis of chemical analyses of soil and groundwater. By using mathematical formulae to estimate the contribution from each exposure route, the daily intake of an individual, or the concentration at an environmental receptor, is estimated. These formulae are submodels and allow adjustment of parameters to the local conditions at a contaminated site. A comparison with, say, acceptable daily intake may then be used as a measure of *risk*. When risks are predicted from chemical analysis of soil and groundwater samples the models can be thought of as running in forward mode. If local conditions are used to calibrate the models they are applied in a procedure generally called *site-specific risk assessment*. *Generic* risk assessment compares concentrations of contaminants with generic guideline values or screening values. These terms often mean the same: a normative value for the concentration of a contaminant in soil or groundwater that is used as a yardstick for preliminary assessment of pollution. In a risk-based approach they are typically the result of models used in reverse mode. That is, for a given risk level (or level for acceptable daily intake), a concentration of the contaminant in soil or groundwater is computed that corresponds with the risk level.

Models are powerful tools for integrating various elements in risk assessment such as site characterisation, fate and transport of contami-

nants, exposure assessment and risk estimation. They are, however, abstract representations of complex systems and are based on numerous assumptions and approximations. It is therefore important that models and submodels are validated and tested in real-world situations, either as part of contaminated land risk assessments or in research projects.

Soil screening and guidelines values are discussed in Chapter 8. Some of these are the results of computer models for risk assessment run in reverse mode. Other values, however, are based on expert judgement and are less explicitly related to risks and adverse effects than the model-based ones. The discussion of soil screening values is included here because they form a link between science-based risk assessment, expert judgement and national policies for contaminated land assessment and remediation.

Chapter 9, 'Better methods for risk assessment' describes the major research gaps identified in the CARACAS programme. These needs are specific for risk assessment of contaminated land. Open problems in the supporting scientific disciplines – for instance, the combined effects of mixtures in human toxicology and ecotoxicology – are not described because they are much more general. Of course, contaminated land risk assessment would benefit greatly from the results of research in these areas. Some of the fundamental science and technology problems relevant to risk assessment are already the subject of active research supported by the European Commission and national governments. One of the important outcomes of the CARACAS initiative is a much better awareness of national and international research programmes, and an enhanced appreciation for the value of international collaboration.

During the discussions in CARACAS it became clear that better risk assessment depends not only on strengthening the research base: successful risk assessment in practice depends on a number of other requirements, which are described under the heading 'Other needs' in Chapter 9.

Further information about CARACAS can be obtained from the CARACAS Web site at http://www.caracas.at. Further information about NICOLE can be obtained from http://www.nicole.org/.

Reference

1. *Towards a Better Future: Establishing Fitness for Use and Sustainable Development of Contaminated Land in Europe.* A joint statement on research needs issued by CARACAS and NICOLE, October 1997.

Chapter 2

Fundamental concepts of risk assessment

2.1 Introduction

Risk assessment is considered a very useful tool in environmental policy because it promises a rational and objective basis for priority setting and decision making. The application of risk assessment to contaminated land problems is widely advocated by many regulators, industries and land developers. Risk assessment methodologies for contaminated land need to be built on a sound scientific basis. This chapter introduces some of the fundamental ideas of risk assessment in the context of contaminated land, and therefore provides a framework for the scientific building blocks considered in later chapters.

A major objective in preparing this book has been to identify research needs so that the scientific basis for risk assessment can be strengthened. The identification of research needs does not imply that using risk assessment in its present state is not worth while. Risk assessment is already a very useful tool in contaminated land management, so long as the users of the method are aware of its limitations. Moreover, sensible use of risk assessment can contribute to the testing of new scientific ideas, which would be much more difficult if current risk assessment methods were not applied in practical decision making.

The origin of risk assessment

The use of risk assessment as a formal component of environmental policy is of relatively recent origin. An original aim of the methodology was to help in setting priorities for environmental protection in an objective and scientific way, thus avoiding conflation with political and management objectives. The highly influential report on the risk

assessment process by the US National Research Council (NRC) describes four distinct stages in the procedure:[1]

- Hazard identification: identification of agents that may cause adverse effects.
- Dose–response relationships: estimating the quantitative relationship between exposure (or dose) and adverse effect from laboratory experiments or epidemiological studies.
- Exposure analysis: estimating the intensity, frequency and duration of exposure to the hazardous agents in question. Depending on context, this can include transport and fate of contaminants in groundwater and surface waters.
- Risk characterisation: evaluation and conclusions that result from the previous steps. In the view of the NRC risk characterisation is a form of expert judgement that should include, to the extent feasible, a description of the distribution of risk in an exposed population.

The distinction between risk assessment (the objective scientific part) and risk management (the policy driven decisions about risks) was central to the NRC risk assessment concept in 1983. However, more recent discussions question the strict separation of assessment from management because specific management problems may determine the way an assessment has to be carried out.

The concept of risk

Although risk is becoming a central concept in environmental policy and practice, this does not mean that it is an easy or well-defined concept. Definitions abound; at least twenty, ranging from informal to very formal (mathematical), were mentioned by Vlek.[2] From the early 1980s there has been an ongoing debate in most developed countries about the measurability or predictability of risk, between scientific and technological risk assessors on the one hand and social scientists and psychologists on the other. The Society for Risk Analysis and its journal *Risk Analysis* continues to be an important forum for this discussion.

These discussions cover the estimation, perception, acceptance and communication of risks of all sorts. Although relevant for the development of risk assessment procedures and risk management decisions for contaminated land, most of this debate about the *nature* of risk does not address contaminated land specifically. The debate can, to a large extent, be characterised by two contrasting points of view: the scientific approach (formal risk assessment) versus the risks as perceived by the general public (intuitive risk assessment). A related dichotomy is that

between objective and subjective inputs to risk-related decision making. Economic risk analysis lies somewhat in between, being formal and highly quantitative and yet based on human preferences. The need for better integration of objective and subjective components is increasingly recognised in almost all fields of risk assessment.

Various types of risk assessment

Risk assessment as a tool is not, of course, limited to the assessment of contaminated land. It is mostly used for other purposes, varying from prevention of pollution from new chemicals and pesticides, through reliability engineering of industrial activities and new technologies, to environmental and financial impact assessment. Recently attempts have been made to link risk assessment of chemicals with life cycle assessment of products. In the context of contaminated land in Europe, risk assessment has mainly been used to set priorities, which means that *risk* is not treated as an absolute quantitative measure describing the environmental or human health impact of soil and groundwater contamination but as an indicator that is used for comparative purposes only. A better term for risk-based priority setting is comparative risk analysis. *Risk assessment* is the scientific process addressing the informal questions 'how risky is it?', or 'what is the chance of a bad outcome?'.

If procedures for risk assessment in different fields are compared, such as exposure to radioactive substances, industrial safety, environmental impact assessment, and risk assessment for contaminated sites, a number of differences may be noted but also a similar basic framework.[3] The most obvious similarities in the various approaches are the universal use of the source–pathway–target concept, and the almost philosophical discussions about the relationship of scientific knowledge, informal judgement and normative issues. The differences concern the choice of endpoint (e.g. life expectancy, death rate, no observed effect concentration), the terminology, societal and political issues about risk (e.g. upper bound of acceptable risk) and the choice of models, parameters and underlying assumptions. Some of these differences are more or less historical, because risk assessment methods have been developed independently in these fields. Reviews covering all types of risk assessment are generally lacking, and exchange of ideas between fields is scarce. Because of the long history of independent development, harmonisation of terminology is unlikely to succeed. However, most terms and ideas are clear enough in the context in which they are used.

Risk assessment of contaminated sites is somewhat different from risk assessment in most other fields. The evaluation of risk from soil contamination is not usually a preventive approach; the source is already there. In principle this makes the assessment easier because claims about exposure can be verified at the site. In practice, however, this advantage is rather limited due to the complexity of the source, the difficulties of performing experiments and (very often) the need to predict *future* exposure. This predictive element means that there is much in common with risk assessment methods used in other fields.

2.2 Current practice

A framework for contaminated land risk assessment

At a general level most countries have a common framework for contaminated land risk assessment procedures. Terminology and matters of detail can vary substantially between countries, which can be quite confusing in international discussions. In some countries *risk from soil contamination* is used almost as a synonym for *pollution*; in other countries *risk assessment* has a very specific meaning and pertains only to a comparison between an exposure estimate and some toxicological limit such as an acceptable intake. Generally the following endpoints are considered:

- Human health: acceptable daily intake (ADI), tolerable daily intake (TDI) and excess lifetime cancer risk are variously used to quantify this endpoint.
- Ecological risk: usually quantified by a no observed effect concentration (NOEC) derived from toxicity experiments.
- Risks to water resources: these risks are related to the relationship between leaching from polluted soils and the dispersion of pollution in groundwater and surface waters. Criteria vary between countries, as do protection levels. In many countries groundwater is protected as a resource that should remain pure. Other countries use risk-based protection levels.
- Construction materials: the effects of soil pollution on structures and construction materials is explicitly considered in the French, Spanish (Basque Country) and UK approaches. In other countries this endpoint is usually implicit.

Risk assessment of contaminated land usually starts with some suspicion about the presence of soil or groundwater pollution. This qualitative information may lead initially to subjective assessment about

environmental risks, and perhaps financial risks for occupiers, buyers, sellers, lenders and others potentially affected by the site. In order to be more certain about the consequences of pollution, a further investigation may be carried out, and contamination levels compared with soil and groundwater criteria. These criteria may be generic or may be derived from site-specific models used to predict adverse effects of pollution and the need for remediation. Model conclusions may be substantiated by actual measurements of exposure or even by epidemiological data. In general, the less prediction the more reliable is the assessment, but the more difficult is the investigation. This trade-off leads to a spectrum of assessment methods with various levels of sophistication. The following assessments are possible:

- History of the site: qualitative expert judgement about likely contamination
- Sampling of soil and groundwater: comparison with generic guideline values or quality standards
- Sampling of soil and groundwater: site-specific modelling of fate, transport and exposure and comparison with toxicological values
- Measurement of exposure or body burden: comparison with toxicological values
- Measurement of exposure: comparison with results from one or more toxicological dose–response models
- Epidemiological study of exposed populations (humans, or other organisms).

A general overview of risk assessment, starting with suspicions and ending with conclusions about risks and how to communicate them, is illustrated in Figure 2.1. Some of the assessment methods mentioned above can be seen as shortcuts in this process. Although risk perception and communication are located at the end of the assessment process in Figure 2.1, it must be stressed that perceptions may influence decision making as soon as suspicions about site contamination are raised. A strength of the risk assessment methodology is that it helps to keep decision making as objective and transparent as possible.

Risk assessment procedures and results need to be accepted by all parties involved. Acceptance in this context involves scientific agreement on methods and the way that they are applied as well as their credibility among non-experts. This raises an obvious dilemma:

- If assessment models are not *black box* but, instead, the assessment steps are made as clear as possible, then risk assessments will be more transparent and comprehensible. Moreover, if methods and models are flexible enough to allow for site-specific adjustment, the

scientific acceptance may be larger than for a rigid application of generic soil screening values.

- Scientific procedures and risk modelling require numerous abstractions of the complex environment being represented. This tends to limit access to a full understanding of risk assessment methods to a few experts. Acceptance by the general public may be more difficult to achieve.

Experience shows that under these circumstances an explicit statement of variabilities, uncertainties and errors is beneficial. By handling variabilities and uncertainties in model applications, their consequences can be studied, risks will be better characterised, and there is a better chance of effective communication with the various stakeholders.

At a general level, the framework for ecological risk assessment may be very similar to that for assessing risks to human health. The target of the assessment, which may be defined as *ecosystem health*, is, however, more complex than human health since it involves a large number of phenomena operating at various spatial and temporal scales. Ecological risk assessment should also make a distinction between two types of ecological risks: (i) those occurring at the contaminated site, and (ii) the impact of the site on the surroundings, either by transport of pollution or by loss of an important habitat. The first type of risk relates to the

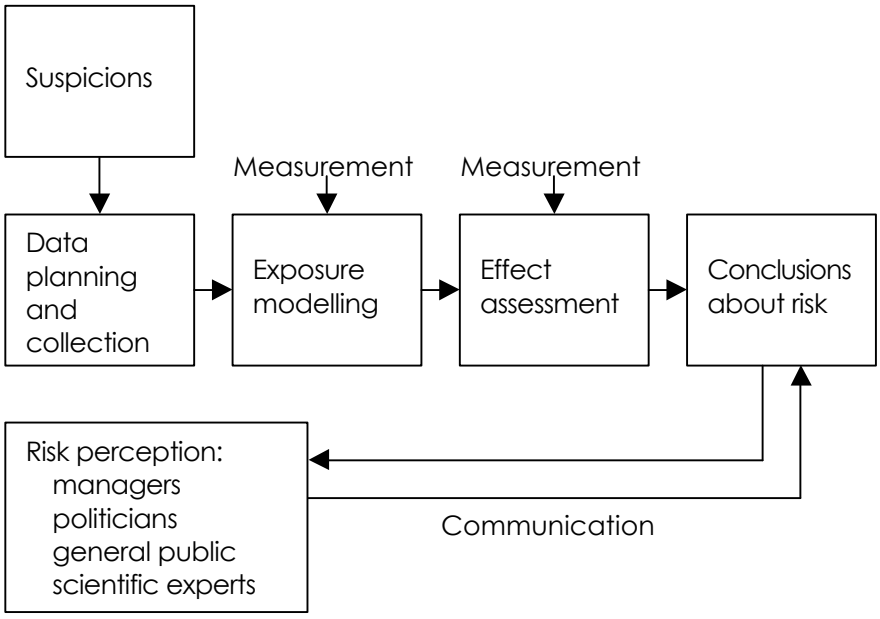

Figure 2.1 The different components of risk assessment. A number of shortcuts are possible as indicated in the text

effects of contamination on the capacity of soil to support life at the site and of water to support the expected ecosystem. The impact of the site on its surroundings may be addressed by procedures quite similar to an environmental impact assessment. Instead of assessing the environmental impact of potential pollution from, say, a new industrial site, the impact of the actual pollution is assessed. However, there is also an important difference between environmental impact assessment and contaminated land risk assessment. The former is used to choose between locations and preventive approaches, whereas the latter addresses the pollution resulting from choices made in the past.

Current use of risk assessment

Some participating countries have policies for contaminated land that do not specify explicit risk assessment procedures, for example Greece. Other countries, such as Portugal and Italy, are starting to develop explicit procedures for assessment and remediation of contaminated land. In countries where such procedures already exist, risk assessment is often used for registering, classifying or prioritising contaminated sites. Experience using risk assessment as a tool to derive site-specific remediation goals is very limited. Even in the Netherlands, probably the European country with the most practical experience of remediation, risk assessment has not been used to derive remediation goals. A simple form of risk assessment was used to classify sites for further investigation under the Soil Protection Act, and to set priorities for remedial action. But solutions for contaminated land problems were framed essentially in civil engineering terms: excavation and replacement of contaminated soil by clean soil, or containment of the pollution by isolation, control and monitoring (ICM) measures. These solutions aim to maximise the control of risk; exposure to soil contaminants is prevented so the difficult debate about *acceptable risk* from pollutants remaining after remediation is circumvented. The Netherlands, in common with many other participating countries, is currently changing its policy towards a more landuse-based approach for the remediation of contaminated soils. This has been prompted not so much by the number of heavily polluted former industrial sites, but by the large-scale diffuse pollution in cities and river sediments. In the new approach there is a much stronger emphasis on the use of risk assessment to specify remediation goals. Experience with this is very limited in the Netherlands, as it is in most other participating countries.

The UK has had a landuse-focused policy for assessing and remediating contaminated land for a long time. This policy is now more

explicitly based on risk considerations than it was in the past. Because most participating countries are now implementing or considering policies that relate the need for remedial action to the actual or future use of the land, there is a growing interest in joint research focused on the relationships between soil pollution, landuse capability, and protection of water resources. It is important to emphasise, therefore, that the terms *suitable for use* and *landuse-based* do not imply an approach narrowly focused on the human activities taking place at a site. The key point is that risks are considered in the context of the specific circumstances of the land in question, and include prevention of water pollution and protection of the wider environment.

In countries with well-developed protocols for assessing contaminated sites, a phased approach is often used. After preliminary assessment the decision to carry out more detailed investigations is usually made on the basis of toxicity and exposure, including environmental fate and transport. Risk assessment is seldom used in the probabilistic sense implied by most formal definitions of risk. If uncertainty analysis were to play a larger role in these decisions, investigations would probably become more costly. On the other hand, there is little point in overly detailed investigation and analysis in situations where conclusions are already clear after a preliminary investigation. Responsible parties (landowners, practitioners etc.) have a choice here, but may still decide to spend more money to increase their level of confidence even if the conclusions seem clear at the preliminary investigation stage.

Priority setting and derivation of remediation goals

For the purpose of setting guideline values (further investigation levels or remediation goals) risk assessment is used in a more rigorous sense than in the priority-setting approach. This calls for a more quantitative and less comparative assessment of risk. Moreover, the aims of the assessment are different. Whereas in priority setting the assessment may establish more or less risky situations, an assessment of risk related to remediation goals must address public safety and environmental quality. Given the uncertainties of risk assessment and the various perceptions of risk by the general public, some countries consider it appropriate to specify upper and lower bounds of acceptable risk in contaminated land policies. For example, the upper bound could be used for priority setting and the lower bound for remediation goals. From a communication point of view, it may be better to frame remediation goals in terms of soil and water quality and fitness for use instead of in terms of risk. In a soil quality approach the soil has to meet

certain physical, chemical and biological requirements specified for different types of landuse, always bearing in mind the need to protect water resources.

Generic criteria versus site-specific risk assessment

There has been considerable debate about whether decision making for risk assessment should be based on generic numerical criteria or on site-specific risk assessment methods. Simple testing of measured concentrations in soil and groundwater against predetermined guideline values is straightforward and less expensive than more elaborate site-specific assessment methods. However, before deciding to invest large amounts of money on remedial action, it may well be expedient to invest in more detailed site-specific risk assessment. A combined approach, using guideline values to streamline the preliminary stages of decision making and site-specific risk assessment to achieve fine-tuning in later stages of an investigation, is generally considered the most appropriate.

Some countries also use predetermined generic soil quality criteria to prescribe remediation goals. In such an approach decisions are made by comparing concentrations of pollutants in soil and groundwater measured at the site with criteria related to the actual or intended use of the site. If these criteria are exceeded remediation is necessary, the remedial objective being to ensure that soil and water quality criteria for the actual or intended land use are no longer exceeded. Use of generic soil quality criteria at every stage of the decision making process, and without scope for flexibility, may be questioned from a scientific point of view. Generic criteria may have significant limitations if they are used as predictors of actual risks in relation to land use. The limitations can be summarised in three main categories:

1. *Exposure period*. Some soil guideline values are based on a lifetime exposure (nominally 70 years), which is traditionally assumed in the derivation of a tolerable daily intake for chronic exposure. However, exposure for 70 years is evidently inappropriate for commercial and industrial sites where the maximum exposure may reasonably be taken as 40 hours per week for 40 working years. On the other hand, the assumption of lifetime exposure may not be considered sufficiently protective if the most sensitive receptors are young children (e.g. because of relatively high soil ingestion rate or high susceptibility to some adverse effects). Both issues indicate the problems that can arise when soil guidelines are used outside the context for which they were derived.
2. *Differences in landuse* lead to obvious differences in human exposure, and hence in potential health risks. Soil pollution, however, has other

potential adverse effects that do not depend on landuse, such as the contamination of water resources. Some ecotoxicological effects might also be important, even when the site is not used as a natural heritage site. For example, adverse impacts on soil microbial processes and vegetation are important even in industrial and urban areas. The range in soil guideline values with respect to land use and human health risks may be too great to provide adequate protection for groundwater, surface waters, and biological processes in soil. This, again, emphasises the need for users of generic soil guidelines to be thoroughly familiar with their derivation.

3. Risk assessment of contaminated land has to deal with a large number of *uncertainties*. Due to the heterogeneity of the soil, and the often capricious nature of contaminative processes, concentrations of pollutants may vary on a very local scale. Consequently, estimates of average concentrations in soil and groundwater may be characterised by wide confidence intervals. The transport and fate of contaminants in soil (including availability and bioavailability) are also variable, which will increase uncertainty about human exposure and risks to water resources. Finally, variability in behaviour, physiology and susceptibility of human beings contributes significantly to the uncertainties in an assessment. In site-specific risk assessment uncertainties are reduced by collecting as much information as possible on the spot. In an approach based on generic soil guidelines, however, site-specific information cannot be used directly. The general assumptions that had to be made in the derivation of criteria of this type could lead to very imprecise statements about health risk from soil pollution. However, this limitation is less severe than it may seem because many of the uncertainties mentioned above are also difficult to take into account in site-specific risk assessment. In practice, both approaches to risk assessment may use uncertainty factors or probabilistic approaches to protect against latent effects arising from uncertainty. In short, if the limitations of generic criteria are taken into account and a conservative approach is used in their derivation, the problems mentioned above may be less important. However, such generic guidelines may look rather stringent.

Where groundwaters are historically polluted, site-specific approaches based on groundwater use are always preferred. For aquifers that have a potential for drinking water abstraction, drinking water standards are often used as the remedial targets. However, where groundwater is not considered to be a usable resource the remedial objective may be related to the quality objectives for surface water bodies into which it discharges. Generic target setting for groundwaters

may lead to overprescriptive remediation since drinking water standards have to be the fall-back comparison. Site-specific consideration also allows for attenuation process to be taken into account along the pathway from the source of groundwater pollution to the chosen environmental target or point of abstraction.

2.3 General methodological aspects

Risk assessment and the management of uncertainties

The scientific definition of risk as *a combination of the consequence of a negative effect and the probability of its occurrence* and the large uncertainties in risk estimation both encourage the use of statistical approaches and the application of decision-support systems. There are several types of uncertainty to be dealt with in a risk assessment and several authors have tried to classify them.[4] For example, Wynne[5] presents the following taxonomy of uncertainty:

- Risk: system behaviour is basically known, and outcomes can be assigned probabilistic values.
- Uncertainty: important system parameters are known, but not the probability distributions.
- Ignorance: what is not known is not known; the degree increases when the level of action or commitment based on what we think we know increases.
- Indeterminacy: causal chains, networks or processes are open, and thus defy prediction.

Shrader-Frechette[6] suggests a number of rules for scientists involved in risk assessment. In the context of environmental risk assessment and decision making, four classes of uncertainty are considered most relevant:

1. Framing uncertainty. This type of uncertainty is related to the translation of policy questions into scientific questions. Does one have to prove beyond reasonable doubt that there is a risk, or that there is no risk? Shrader-Frechette advocates the use of a so-called three valued frame: (i) there is no risk, or (ii) the decision is not possible due to lack of information, or (iii) there is a risk.
2. Modelling uncertainty. This type of uncertainty pertains to the realism of models, and to the question of the reliability of model predictions. Very often models are considered to be validated or verified if the output of the model is consistent with some other

model. The only valid test is the comparison of model predictions with real-world data. If real-world phenomena are successfully predicted one might gain confidence in a model, but asserting that a model is true on the basis of successful predictions is scientifically speaking not possible. In science empirical data may lead to falsification of hypotheses but not to their confirmation. On the other hand, a model that is well corroborated (i.e. has survived many attempts at falsification) can be used with more confidence than one that is not.

3. Statistical uncertainty. This refers to the so-called type I and type II errors in statistical analysis. Scientists tend by tradition to minimise type I errors, the chance of rejecting a true hypothesis. Type II errors, the chance of accepting a false hypothesis, may be more serious in environmental problems, especially when the hypothesis was that there are no health risks or ecological risks. For this reason it is preferable to pose a null hypothesis in terms of there is a risk rather than there is not a risk.
4. Decision theoretic uncertainty. This type of uncertainty shows up in risk-related decisions. Should the worst case scenario govern the decision even if it has a very low probability of occurrence? Or must the decision be based on the more likely scenarios? Or should decisions be based on utilitarian principles and cost–benefit analyses? How should site risk assessments be modified to take account of high background exposures from non-site and non-soil sources of contamination?

The considerations outlined above may lead to the following rationale in risk assessment for contaminated sites. Risk assessment can be used in two ways: to estimate the risks at a given site in order to establish the necessity and priority of remediation; and to set generic or site specific remedial goals, and to predict residual risks after (partial) remediation. It is well understood that risk assessment is subject to large uncertainties especially when it relies extensively on modelling. Uncertainties can be reduced if model predictions can be corroborated and refined by additional measurements. However, many risk assessments relate to future situations, and are inevitably associated with large uncertainties. In principle if risk assessment is used to derive remedial goals, larger uncertainty factors are usually considered to be appropriate.

The perception of risk

It is well known that the results of formal scientific approaches to risk assessment can be very different from the risks as perceived by the

general public. Risk perception may be governed by a number of factors, but the main difference is that risk is perceived more intuitively. Risks related to adverse effects that are conspicuous, known from experience, or that occurred recently or in the immediate surroundings are often overestimated. Another factor in perceived risk is the neglect of initial probabilities of certain phenomena: a warm summer may be seen as proof of global warming even if the temperature is consistent with normal fluctuations. A third factor is that intuitive estimates of probability are sometimes based on analogy or inspired by estimates of other chance phenomena.

Apart from bias in our capacity to assess certain risks, risk perception is also influenced by personality factors. Some people are more optimistic than others. Optimistic people might underestimate risks whereas pessimistic people tend to overestimate them. On the other hand, pessimistic people may be more analytical and assess risks more according to formal scientific models. Another factor is that people are not likely to change their opinions about risk. Perception is persistent.

In general people find the estimated probability of an event less influential than the nature of the outcome. This is important for the acceptance of the risk. Acceptance is less if the event:

- is a catastrophe
- affects a large area
- is not controllable
- is new, or outside previous experience
- leads to unfair distribution of advantages and disadvantages
- is against one's will
- is dreadful
- is imminent rather than far into the future.

The factors influencing risk perception and acceptance are important for communicating the results of formal risk assessment to the general public and other stakeholders, and for negotiating effective risk management. Effective communication depends not only on public perception but also on the characteristics of the communicator, the message and the medium that is used. It is beyond the scope of this chapter to give a full treatment of risk communication. One final suggestion is that it may be easier to communicate about soil quality (*this piece of land is still fit for ... and will be fit for ... after remedial action*) than to communicate about risk (*there is still pollution left but the authorities state that your risk is acceptable*).

Risk assessment and risk management

Recent discussions question the strict separation of risk assessment (the objective scientific part) and risk management (the policy driven decisions about risks) for a number of reasons:

1. Politicians have to take the perceptions of non-experts into account. Public perception of risk might frame the questions to be asked during an assessment. A formal scientific assessment may not fully answer the political questions.
2. If scientific assessments are made very detailed and very specific, they almost dictate the decision to the risk manager. On the one hand this simplifies and accelerates the decision making; on the other hand the autonomous role of the decision maker is reduced. This trade-off calls for a dynamic interplay between assessment and management of risk.
3. Whether a certain risk is acceptable is not a scientific question. Scientists can assess the likelihood of occurrence of an adverse effect. However, if they act according to the scientific tradition that there is no effect unless it is proven beyond all statistical doubt, risk management will not be very protective. According to Shrader-Frechette,[6] scientists involved in risk assessment must avoid false positive conclusions about risk as well as false negative ones. Risk assessment needs a *three-valued frame*: (i) negligible risk; (ii) risk cannot be proved to be absent or present; (iii) risk is present beyond reasonable doubt. This leads to another non-scientific question in risk assessment: how much doubt is reasonable?
4. Risk assessments can involve statements about complicated and poorly understood phenomena. Even for situations with adequate dose–effect relationships, a no effect level may be estimated but cannot be verified in practice. To detect low levels of risk very large sample sizes are needed. Scientific proof is not possible in practice. This has been labelled a *trans-scientific* question by Weinberg.[7]
5. Some people argue that if risk assessment is so plagued by lack of information, large uncertainties and numerous trans-scientific questions, then it cannot be a very powerful tool. Other people might say: it is the best we have because it is the only way to make objective decisions. Sceptics may say: wrong decisions are wrong even if objective! Risk assessment is also seen by some as a cosmetic activity intended to give a false sense of certainty or answerability, or as a kind of expert mumbo-jumbo used to delay risk reduction actions. According to O´Brien[8] scientists should move away from the hubris of assimilative capacity estimation and risk assessment to the

wisdom of a precautionary orientation. One might also say that if scientific risk assessment is so difficult, the more intuitive political perception of risk may provide better guidance in decision making. Risk assessors recognise that there is a grain of truth in each of these extreme positions, and that they all play a part, to a certain degree, in practical risk assessment and decision making.

Nevertheless, if risk assessment and risk management cannot be separated (as these arguments suggest) an improvement in the scientific basis for risk assessment is of the utmost importance for the whole decision making process. What seems clear is that risk assessment is not mature as a scientific discipline, and has not yet gained public trust in the way that more established disciplines have:

'When it comes to judging a risk, most people would rather trust the opinion of a friend than take the word of a scientist.'

(*New Scientist*, 28 September 1996)

2.4 Concluding remarks

Risk assessment, risk analysis, policy making and decision making are extensively studied only in the social sciences and psychology. Apart from the ongoing debate between the scientific–technical and sociological–psychological points of view, there are no clear trends that are specific for contaminated land risk assessment. Attempts[2,9] to integrate the technical framework with the socio-psychological aspects in a decision-theoretic approach might yield valuable results in the future.

Risk assessment for contaminated sites is still a rather loose assemblage of concepts and methods borrowed from various scientific disciplines. Further integration may be achieved under pressure of environmental policy; by closer dialogue between governments, industry and academic researchers; and by providing funds for special research and development programmes. It is doubtful whether contaminated land risk assessment would emerge as an integrated scientific discipline on its own through autonomous developments in existing scientific disciplines.

To make risk assessment more useful for policy decisions it must fit in with the various national policies for contaminated land. Policy considerations frame the questions to be addressed in risk assessment and delimit the borderline where science ends and policy begins. If, for instance, an ADI (Acceptable Daily Intake as published by the World Health Organization) is accepted as a toxicological limit value by policy

makers, risk assessments may estimate the chance of exceeding the ADI. If the ADI is not accepted and a policy statement like *substantial health risk should be avoided* is the only basis for risk assessment, the poorly specified relation between daily intake and human health has to be incorporated somehow into the scientific risk assessment process.

Some participating countries do not have policies for contaminated land that specify explicit risk assessment procedures. Other countries are starting to develop formal procedures for assessment and remediation of contaminated land. In countries where such procedures already exist, risk assessment is mostly used for registering, classifying and prioritising contaminated sites. Experience with risk assessment as a tool to derive site-specific remediation goals is very limited. Most countries are now considering policies that relate remediation needs to the actual or future use of the land, and want to use risk assessment in decision making about remedial actions. Three objectives in risk management policy that most countries would be happy to endorse in principle can be used as focal points in RTD programs. They are:

1. Fitness for use: contaminated land may be unfit for its present or intended use. A *fitness for use assessment* will have to assess the risks to humans, water resources, ecosystems and other environmental endpoints as related to the use of the land. Joint research at the European level would benefit by being focused on the relationships between soil pollution and landuse capability.
2. Aftercare: in general, more clean up now will mean less aftercare in the future, although this may mean greater costs now. However, greater insight into the long-term behaviour of soil contamination, e.g. biodegradation, and transport and fate of pollutants in groundwater, will be important in assessing the need for future aftercare.
3. Protection of the environment: even if the soil and water quality at a site make it fit for its current use in the narrow sense, a site may still pollute other parts of the environment; for example, secondary poisoning of bird species that occasionally forage there. The policy goals for environmental protection (for instance, the EU Groundwater Directive) and the policy goals for contaminated land remediation converge here. Research into how to predict the impact of a site on the surrounding environment is therefore of central importance to risk assessment.

References

1. NRC (National Research Council) (1983) *Risk Assessment in the Federal Government: Managing the Process*. National Academy Press, Washington, D.C.
2. Vlek, C.A.J (1990) Beslissen over risicoacceptatie. *Gezondheidsraad* A 90/10, Den Haag.
3. Moen, J.E.T., Jansen, M.P.M., Slaper, H. and Lembrechts, J.F.M.N. (1994) Risicovergelijking tussen starling/stoffen, eerste tussenrapport. RIVM no. 610 052 001, Bilthoven.
4. Wynne, B. (1972) Uncertainty and environmental learning: reconceiving science and policy in the preventive paradigm. *Global Environ. Change* **2**, 111–127.
5. Dovers, S.R. and Handmer, J.W (1995) Ignorance, the precautionary principle and sustainability. *Ambio* **24**(2), 92–97.
6. Shrader-Frechette, K. (1996) Methodological rules for four classes of scientific uncertainty. In: *Scientific Uncertainty and Environmental Problem Solving* (J. Lemmons, ed.). Blackwell Science, Oxford (ISBN 0-86542-476-4).
7. Weinberg, A.M (1972) Science and trans-science. *Minerva X* **2**, 202–222.
8. O´Brien, M.H (1994) The scientific imperative to move society beyond 'just not quite fatal'. *The Environmental Professional* **16**, 356–365.
9. Royal Society (1992) *Risk: Analysis, Perception and Management*. The Royal Society, London (ISBN 0-85403-467-6).

Chapter 3

Receptors: human health

3.1 Introduction

Human health risk assessment of contaminated sites is often subdivided into four phases: hazard identification; dose (concentration)–response (effect) assessment; exposure assessment; and risk characterisation. Assessing the impact of soil contaminants on human health therefore requires reliable data on human exposure and the resulting possible adverse effects. This chapter reviews the practical application of human toxicology studies in the context of contaminated land risk assessment. It also attempts to summarise some of the variations in practice between various participating countries. The focus is on addressing the following questions:

- What are the criteria for selecting priority contaminants?
- What are the criteria for selecting appropriate toxicological/epidemiological studies for use in contaminated land risk assessment?
- How are the toxicology studies interpreted?
- How are the results applied?

It is important to recognise that much of the toxicology data used in contaminated land risk assessment were derived from animal experiments performed for different purposes, such as establishing acceptable daily intakes of additives and environmental contaminants in food. Usually, the chemicals to be tested were administered orally in the feed or by gavage in a pure and soluble form. Thus the chemical form, mode of delivery and exposure conditions are not necessarily appropriate for contaminated land risk assessment. Consequently, most permissible intake values are probably very conservative in the context of exposure to soil contaminants.

3.2 Criteria for selecting priority substances

The list of hazardous substances that might be found on a contaminated site could be potentially almost without limit. Many countries have therefore selected priority substances for contaminated land risk assessment. These may be substances for which soil guideline or target values are published, for which toxicology and other data are compiled for ease of reference, or which are *priority substances* in some other sense. The main factors used in selecting priority substances are listed below:

1. Human toxicity:
   (i) systemic toxicity (acute and chronic effects)
   (ii) carcinogenicity and genotoxicity
   (iii) reproductive and developmental effects
   (iv) neurobehavioural toxicity
   (v) local effects (e.g. irritation)
2. Likely presence of the substance in significant concentrations on land affected by past or current industrial use
3. Toxicity to plants and animals
4. Potential for bioaccumulation and biomagnification
5. Mobility in the environment (e.g. solubility, volatilisation potential)
6. Persistence in the environment and residence time in soils
7. Potential to explode or ignite
8. Potential to damage or impair performance of building materials

Most participating countries that have produced a priority substances list appear to have used some or all of the above factors in a structured way, although the relative importance of the factors will of course vary from substance to substance.

3.3 Criteria for selecting appropriate toxicological and epidemiological studies

Certain chemicals and commercial products, e.g. pesticides, food additives and pharmaceuticals, are required to undergo toxicity testing according to strict protocols such as those set out by the OECD[1] and WHO.[2] The objective is to establish tolerable ('safe') human dosage or exposure levels before approval by national or international agencies. The tests are primarily designed to protect human health; environmental effects may or may not be considered.

Standard toxicity studies include:

- Acute toxicity – single dose
- Subacute toxicity – repeated administration for 14–28 days
- Subchronic toxicity – repeated administration for (usually) 90 days
- Chronic toxicity – lifelong administration
- Carcinogenicity – lifelong administration
- Teratogenicity – administration during gestation
- Reproductive effects – multigeneration administration
- Genotoxicity – *in vivo* and *in vitro* test systems.

Additional testing may include:

- Toxicokinetics
- Sensitisation
- Skin and eye irritation
- Neurotoxicity.

Most environmental chemicals, including soil contaminant mixtures and waste and combustion products, have not been adequately tested for toxicity in a systematic way. Information on toxicity has to be deduced from published studies on effects on humans, experimental animals or other organisms.

The major sources of compiled information used in most participating countries appear to be:

- World Health Organization/International Program on Chemical Safety (WHO/IPCS) Environmental Health Criteria documents
- WHO Guidelines for Drinking Water Quality and Air Quality in Europe
- USEPA Integrated Risk Information Service (IRIS)
- Monographs of the International Agency for Research on Cancer (IARC)
- Reports of the Agency for Toxic Substances and Disease Registry (ATSDR).

However, Tolerable Resorbed Dose (TRD) values used in Germany have been developed for the specific purpose of human health risk assessment in the context of contaminated land.[3,4] In the UK relevant reviews of the Committees on Toxicology (COT) and Carcinogenicity (COC) are considered authoritative.

In many published scientific studies, detailed information on dose–response relationships are lacking and the quality of the data may be uncertain. Moreover, these studies rarely address the question of combined exposure to mixtures of chemicals. The selection of appropriate toxicity studies for risk assessment largely depends on the expert judgement of the toxicologist or responsible risk manager.

3.4 Interpretation of toxicity data

The interpretation of available toxicology information for use in human health risk assessment of chemicals is complex. Data from epidemiological studies involving groups representing the general population and whose exposure level is well characterised would be the ideal situation. Data from occupationally exposed groups also provides valuable information, although the exposure levels, exposure duration and the sensitivity between individuals within this group may be different from those of the general population. Moreover, workplace exposure frequently involves mixtures of chemicals. Epidemiological studies are generally rather insensitive, and subtle changes in disease incidence are likely to remain undetected.

Despite these difficulties, most participating countries prefer to use epidemiological data if possible, although the limitations of such data are well recognised. Examples of tolerable intakes derived from human studies include arsenic, benzene, cadmium and free cyanide.

Even when human data are available, exposure levels are often poorly constrained. Information on dose–response relationships is therefore largely derived from animal experiments. However, the exposure levels used in animal testing may be several orders of magnitude higher than the expected human exposure levels. This is because the incidence of biological effects needs to be sufficiently high to be detectable in a group consisting of a limited number of animals.

Use of animal studies

Two procedures are frequently used to estimate the risks to humans at exposure levels lower than those used in experimental studies. The first approach is the use of uncertainty (or safety) factors applied to the highest exposure level at which no adverse effects are observed (the No Observed Adverse Effect Level; NOAEL). The NOAEL can be deduced from animal experiments or human observations. This is the approach used by many agencies to estimate the tolerable daily intake (TDI) for non-genotoxic chemicals.

The uncertainty factors typically used as a starting point in most countries are summarised below; these largely follow the WHO convention:

- interspecies variation (laboratory animal to man): factor of 10
- intraspecies variations (sensitive individuals): factor of 10

- use of Lowest Observed Adverse Effect Level (LOAEL) rather than NOAEL: factor of 10
- subchronic rather than chronic study: factor of 10
- poor database: variable factor.

These are not necessarily applied rigidly, and some may be combined (e.g. in Denmark the last three are combined into a security factor, SF_3, which depends on the quality and relevance of the available studies, and which may vary from 1 to 100).

The second approach involves mathematical modelling of a toxic response as a function of dose. The approach of choice depends on the mechanism of action of the chemical in question. For all chemicals except genotoxic substances, a threshold level of exposure is assumed, below which there is no significant risk of adverse effects.

An example of the mathematical modelling approach in dose–response assessment is the recent development of the benchmark dose (BMD) concept as an alternative to identifying a NOAEL and dividing this by uncertainty factors. The BMD is the dose that corresponds to a specified level of increased response (called the benchmark response) typically set at the lower end of the range of responses that can be detected experimentally.[5,6] This approach makes better use of dose–response information and also reflects sample size more appropriately than the NOAEL approach. Furthermore, a BMD is not constrained to be one of the experimental doses and can be defined from a data set in which there is no NOAEL. At present none of the participating countries uses the BMD approach for the derivation of guideline values (mainly because it has not yet been adopted by major agencies such as WHO and USEPA) although it has been used in site-specific risk assessments in some countries.

When data from experimental animals are used, interpretation of the results for use in human risk assessment raises several questions, such as the appropriateness of the animal model used, exposure route, dosage regimen, duration of exposure, age and gender differences, statistical treatment of data etc. Studies on lower organisms and results from *in vitro* studies may yield valuable additional information regarding mechanisms of toxicity but the quantitative data are generally not applicable to human exposure. In some instances, risk assessment may involve consideration of structural or chemical similarities between known toxic compounds and suspected agents. In addition, some risk assessors try to incorporate biological models, such as the physiologically-based pharmacokinetic (PBPK) family of models, into decision making. These models quantitatively take account of the differences between laboratory animals and humans by considering body

weight, surface area, metabolic capacity and products, blood flow, respiration rate, body fat contact and several other factors. An example is the work of Johanson and Filser[7] on butadiene. Paustenbach[8] lists some 40 widely used chemicals for which PBPK models have been developed. PBPK models also offer potential for studying the effects of mixtures of chemicals.[9]

In Europe, in the context of contaminated land risk assessment, there appears to be little explicit use of PBPK models and structure–activity relationships to establish toxicological criteria. The exception is for lead, where the USEPA's IEUBK model is used in some countries (e.g. Belgium) to derive guideline values. In other countries information derived from these models may be used as part of the overall weight of evidence favouring one study or approach over another.

Carcinogenic substances

Depending on the mode of action, carcinogenic chemicals may be divided into genotoxic carcinogens (mutagens) and non-genotoxic carcinogens (non-mutagens). This distinction can have repercussions on risk assessment in that mutagens are usually regarded as not having a no-effect level; that is, a permanent generic alteration in a single cell is theoretically sufficent to induce a critical mutation which may lead to a transformed cell progeny, and eventually to tumour initiation. In contrast, non-mutagens may show a threshold level of exposure below which biological effects are negligible. When no threshold is assumed, an exposure level at which the health risk is assumed to be sufficiently low to gain general acceptance is extrapolated or calculated.

Most participating countries appear to use extrapolation models favoured by the WHO and USEPA (e.g. *one-hit* and *linear multistage*), or use directly the slope factors or unit risks published by these agencies.

Extrapolation using slope factors is not endorsed by the UK Committee on Carcinogenicity as a method for routine use. The reasons given are:

- the extrapolation models are not validated;
- they are often based on incomplete or inappropriate data;
- they are derived more from mathematical assumptions than knowledge of biological mechanisms.

It is preferred to use a combination of information from animal experiments, epidemiology and knowledge of biological mechanisms.

The particular weight given to the various aspects will vary from substance to substance.

For chemicals that appear to contribute to increased cancer incidence through non-genotoxic mechanisms and where a threshold is therefore likely, NOAELs and LOAELs with uncertainty factors may be used by regulatory agencies. The Institute of Environmental Medicine in Sweden, for example, has used an uncertainty factor of 5000 for some non-genotoxic carcinogens. The factor takes into account interspecies and intraspecies extrapolation (10×10) and the severity of the effect (10), and introduces an additional factor (5) when appropriate to account for extrapolation from LOAEL to NOAEL.

The distinction between genotoxic (non-threshold) and non-genotoxic (threshold) carcinogens in risk assessment is controversial. The USEPA has, for instance, generally used mathematical (slope factor) modelling also for non-genotoxic carcinogens whereas the WHO has advocated the uncertainty factor approach. Recently, the USEPA[10] has proposed evaluation of carcinogenic chemicals on a case-by-case basis, considering the most plausible biological mechanism of action. Most participating countries already make a practical distinction between genotoxic and non-genotoxic carcinogens whenever the data allow.

The theoretical tolerable excess lifetime cancer risk typically used in the context of genotoxic carcinogens on contaminated sites ranges from 10^{-6} (e.g. Denmark) to 10^{-4} per substance (Netherlands), with the majority of countries preferring 10^{-5}. In the UK theoretical lifetime risks around 10^{-5}–10^{-4} are generally considered acceptable for the general public, bearing in mind that the real risk (albeit unknowable) is unlikely to be higher and may well be very much lower. However, derivation of tolerable daily intakes by dividing maximum tolerable risk by slope factor is not generally favoured in the UK.

In the UK carcinogens are treated as additive, i.e. the ratio *estimated intake/TDI* for each carcinogen in a sample is calculated and the ratios added. When the sum exceeds unity the TDI for carcinogenic substances is considered to be exceeded. In Germany the maximum tolerable lifetime excess cancer risk for individual substance is set at 10^{-5}, but the total excess risk from all carcinogens combined should not exceed 5×10^{-5}.

Despite the scope for misunderstanding the idea of theoretical excess cancer risk, none of the participating countries appears to provide explicit guidance to the various stakeholders (including the general public) on how the numbers should be interpreted.

3.5 Application of results

Bioavailability

Some of the most fundamental problems for contaminated land risk assessment relate to bioavailability of contaminants relative to the study or studies on which the toxicity criteria were based. In particular, will soil-bound contaminants be less bioavailable and hence less readily absorbed from the gut than those ingested with food and water? Several distinct issues can be identified, some of which are equally relevant for understanding the transport and fate of contaminants in the environment (see Chapter 6).

- **Mineral speciation**: some toxic metals are found as components of relatively insoluble naturally occurring minerals (e.g. lead in pyromorphite and galena; cadmium in sphalerite). Thus, these metals may be much less bioavailable in mining communities than, say, in smelter communities or urban areas.[11]
- **Chemical speciation**: toxicity criteria are often based on experiments using pure compounds in a soluble form. The same substance in soil may occur in a variety of chemical species. For example, nickel may occur as free cations, sulphates, sulphides, phosphates, ferrites, carbonates, bicarbonates and hydroxy- complexes. Nickel can also be mobilised by chelation with organic contaminants, even in soils with an apparently high binding capacity.
- **Sorption effects**: contaminants in soil are often strongly adsorbed onto the clay, ferric hydroxide and/or organic matter fractions of soil. The soil may thus act as a sink for contaminants. Organic contaminants that have been in the soil for many years may be much less bioavailable than their newly introduced equivalents. This reflects the combination of processes sometimes referred to as *ageing*, which includes strong surface adsorption, partitioning into soil organic matter and diffusion into micropores.[12]
- **Transit time**: in contrast to factors that reduce bioavailability, it needs to be recognised that soil may have a longer residence time in the gut because it becomes entrained by the gut mucosa.[13] Indeed, soil is sometimes used as an animal food amendment to increase retention time and hence maximise nutrient uptake.[14]
- **Experimental studies**: some aspects of relative bioavailability can be studied using *in vitro* enzymolysis, for example of animal feed to which various amounts of soil are added. *In vivo* experiments are also possible using either radiotracer additions to rodent food, alone

or with an added proportion of soil, or by directly dosing animals with samples of contaminated soil by gavage or as a soil–food mixture.

In the context of deriving national soil target values, relative bioavailability may be taken into account in various ways, usually considered on a case by case basis:

- The interspecies uncertainty factor covers, among other things, toxicokinetic differences such as relative absorption in the gastrointestinal tract between experimental animal and man.
- The availability of some soil contaminants for uptake by edible plants has been studied experimentally.
- Chemical analysis may be based on *extractable* concentration rather than *total* concentration.
- A *reduction factor* may be introduced into exposure models (e.g. the Dutch CSOIL model).

However, because there is so little information on absorption from a soil matrix relative to the chemical forms and vehicles used in animal studies, most countries take the precautionary view that the soil matrix does not significantly reduce bioavailability.

Similarly, although technical guidance in various countries includes discussion of relative bioavailability, it is generally left to risk assessors to make their own judgements in individual cases. To provide support in Germany and the Netherlands, *in vitro* methods are being developed which should provide a sounder basis for expert judgement.

Routes of entry

For many contaminants adequate toxicological studies exist for only one route of entry, usually ingestion. If intake also occurs by other routes (inhalation, dermal) it may be necessary to consider the absorption efficiency relative to absorption via the gastrointestinal tract. This procedure (adjustment for relative absorption efficiency) is only appropriate where the toxicological endpoint is the same irrespective of route of entry.

Most national soil guideline values are derived on the basis of combined exposure via ingestion and inhalation (and sometimes by dermal absorption also). Intakes are not usually adjusted for relative absorption, usually because there are insufficient data for most contaminants.

For other contaminants, the target organ differs depending on route of entry. With nickel, for example, the critical endpoints for exposure by

inhalation are cancer of the lungs and nasal passages. In contrast nickel does not appear to be a human carcinogen via ingestion. It is, however, a potent skin sensitiser, therefore ingested nickel can contribute to nickel dermatitis. When toxicological endpoints depend on route of entry, soil target values are typically based on the most sensitive end-point.

Recent research suggests that because fine-grained particles will preferentially adhere to the skin, the contaminant concentration of adhering soil will be greater than that in the bulk soil. This enrichment factor may be substantial (e.g. around 10 for metals in sandy soil) although there appear to be few data, if any, for organic contaminants. Enrichment as a function of grain size and/or organic matter context may need to be considered for the ingestion (hand-to-mouth transfer), dermal and vegetable consumption pathways.

In the UK default enrichment factors for metal contaminants in the context of dust inhalation are taken as 1.5 for clay soils, 3.0 for loam soils and 10 for sandy soils, based on the work of Sheppard and Evenden.[15] In Germany a default enrichment factor of 10 is used for inorganic contaminants and 5 for organics. Other participating countries do not appear to incorporate enrichment factors in their derivation of soil guideline values.

Exposure to chemical mixtures

Simultaneous exposure to mixtures of chemicals is a general problem in environmental risk assessment. Multiple exposure can give independent, additive (or non-interactive), synergistic or antagonistic effects. Given the vast number of potential combinations of hazardous substances, there is little prospect of a systematic experimental approach to mixtures toxicity. Structure–activity relationships and PBPK modelling to assess uptake, distribution and disposition of compounds in humans, may offer a partial solution to the problem. At present few participating countries give explicit guidance on assessing risks from mixtures of contaminants, although the WHO viewpoint is considered authoritative in many countries.

For substances with a dose threshold, the WHO position[16] in the context of drinking water quality guidelines is that guidelines are:

> 'calculated separately for individual substances, without specific consideration of the potential for interaction of each substance with other compounds present. However, the large margin of safety incorporated in the majority of guideline values is considered

sufficient to account for such potential interactions. In addition, the majority of contaminants will not be present at concentrations at or near the guideline value.'

However, the text goes on to say that there may be:

'occasions when a number of contaminants with similar toxicological effects are present at levels near their respective guideline values. In such cases, decisions concerning appropriate action should be made, taking into account local circumstances. Unless there is evidence to the contrary, it is appropriate to assume that the toxic effects of these compounds are additive.'

This leaves the decision making rather open-ended. How similar do the *similar toxicological effects* need to be? Should only those effects upon which the NOAEL is based be taken into account? How near to the respective guideline values should the concentrations be before applying the additive rule?

At the other extreme, for many years the USEPA advocated use of a *hazard index* in which each contaminant intake was expressed as a fraction of the reference dose (=TDI), with the fractions then added. This implies strict additivity even for contaminants that affect different tissues or organs by different mechanisms. It was recognised[17] that:

'the assumption of dose additivity is most properly applied to compounds that induce the same effect by the same mechanism of action. Consequently, the application of the hazard index equation ..., although appropriate as a screening-level approach, could over-estimate the potential for effects.'

The UK Health and Safety Executive,[18] in the context of workplace exposure to mixtures, takes an intermediate position. It recommends a hazard-index type of approach for substances that are known to have additive effects but otherwise it is sufficient to ensure compliance with each exposure limit individually. However, it then points out that:

'It is open to people responsible for control of exposure to treat all non-synergistic systems as though they were additive. This avoids the need to distinguish additive and independent systems and can be regarded as the more prudent course, particularly where the toxicity data are scarce or difficult to assess.'

In the Netherlands contaminant mixtures are treated as fully additive within restricted chemical groups, e.g. Cd + Pb + Hg; aldrin, dieldrin and related pesticides; chlorophenols; chlorobenzenes.

Toxicity of some major groups of structurally-related compounds (e.g. dioxins and furans, PAHs) may be assessed using a toxicity equivalent factor (TEF) approach. This is also a form of effects additivity, although the TEFs are sometimes little more than educated guesswork. The TEF approach is used in Denmark, the Netherlands and the UK for PAHs, dioxins and dioxin-like PCBs, and for dioxins and dioxin-like PCBs in Sweden. In Germany the approach is used for dioxins and is being considered for PAHs. In Norway consideration is being given to the TEF approach but no decision has yet been made.

Exposure from multiple sources

Most human health risk assessments for contaminated sites are based on estimating the contaminant concentration in soil that would give rise to the maximum tolerable intake via ingestion, inhalation and dermal absorption. This is uncontentious when site soil is the only significant contaminant source. However, for many environmental contaminants, people are also exposed via other non-site and non-soil sources such as traffic, diet and household chemicals.

In some countries (e.g. Belgium and the UK) estimated intake from non-soil sources (here called background intake, BI) is subtracted from the tolerable daily intake (TDI) to give a tolerable daily intake from soil (TDSI = TDI – BI). However, as this difference gets smaller soil target values based on TDSI become increasingly stringent and the cost of achieving the target intake for soil would be grossly disproportional to the contribution that site soil makes to total risk.

In most other countries BIs are ignored when setting guideline values but are meant to be taken into account in site-specific risk assessments. Some countries respond in a third way – by setting the site contribution to total risk at an arbitrary percentage. This approach is followed in Denmark (10% allocated to soil exposure) and Germany (20% of total resorbed dose for soil sources is the default assumption). In the UK 50% of the TDSI is typically allocated to on-site soil; the remaining 50% is to allow for exposure to soil-borne contamination at other localities in a neighbourhood. However, for genotoxic carcinogens the on-site soil allocation is set to 100% and BI is ignored; this is to ensure that the focus is on reducing excess risk from a site without regard for other carcinogens in the environment.

It can be important to distinguish between toxic effects that depend on route of entry into the body. For example, chromium VI is carcinogenic by inhalation but appears not to be carcinogenic by ingestion. In general, tolerable intakes and slope factors derived from one route of

entry should not be used for other routes of entry. In practice, however, for many substances reliable information on toxicity may only exist for one route of entry. It is then common practice to compare such a toxicity guideline with total intake, or with uptake if sufficient information on relative absorption via different routes of entry exist.

3.6 Summary and conclusions

This chapter has addressed four main problems: the criteria for selecting priority substances for contaminated site risk assessments; the criteria for selecting appropriate toxicological and epidemiological studies; how the toxicity studies should be interpreted; and how the results are applied.

Priority substances are usually selected mainly on the basis of human toxicity and the likely presence of the substance in significant concentrations on land affected by past and current industrial use. There is a wide range of toxicity tests relevant to human health protection. Most participating countries use reports published by the major agencies (WHO/IPCS, USEPA, IARC, ATSDR) as sources of compiled information on human toxicity.

Tolerable daily intakes derived from human studies are relatively rare. Information on dose–response relationships is therefore largely derived from the results of animal experiments, in conjunction with the use of more or less conventional uncertainty factors. Interpretation of animal studies in the context of human health risk assessment raises questions about the appropriateness of the animal model or models used, the exposure route and duration of exposure, the dosage regime, and the statistical treatment of data.

In the context of exposure to genotoxic soil contaminants, most participating countries set a theoretical tolerable excess lifetime cancer risk in the range 10^{-6} to 10^{-4} per substance. However, basing a tolerable daily intake on a maximum tolerable risk divided by a cancer potency slope derived from animal experiments is not favoured by all participating countries.

It is widely recognised that soil-bound chemicals may be significantly less bioavailable than those used in the study or studies on which the tolerable daily intake was based. Factors that differ include mineral and chemical speciation, sorption effects and residence time in the body. *In vivo* and *in vitro* experiments are being conducted to provide a sounder basis for expert judgement on relative bioavailability.

Simultaneous exposure to mixtures of chemical is a general problem in environmental risk assessment. Most participating countries appear

to be guided by the WHO's position on exposure to mixtures. For some major groups of structurally-related compounds, such as dioxins–furans and PAHs, the toxicity of mixtures is treated in several countries by the use of toxic equivalency factors.

Some countries take explicit account of contaminant intake from background (non-site) sources when deriving soil guideline values, or in site-specific risk assessment. Other countries provide protection against non-site exposure by setting the site contribution to an arbitrary fraction of the total tolerable exposure or risk. There is widespread agreement that the management response to soil contamination should in some way be proportional to the contribution that soil sources make to the total risk from the relevant contaminants. However, the decision-support tools for helping to take these complex issues into account are as yet poorly developed.

References

1. OECD (1981) *Guidelines for Testing of Chemicals*. Organization for Economic Cooperation and Development, Paris.
2. WHO (1987) Principles for the safety assessment of food additives and contaminants in food. *Environmental Health Criteria* no. 70. World Health Organization, Geneva.
3. Hassamer, M., Kalberlah, F., Oltmanns, J. and Schneider, K. (1993) Basisdaten Toxikologie für umweltrelevante Stoffe zur Gefahren-beurteilung bei Altlasten. *Umweltbundesamt Berichte* 4/93. Erich Schmidt Verlag, Berlin.
4. Kalberlah, F., Hassamer, M., Frijus-Plessen, N. and Schneider, K. (1996) Toxicological risk assessment of soil contaminants. Presented at the International Symposium on Exposure and Risk Assessment with Respect to Contaminated Soil, GSF – Forschungzentrum für Umwelt und Gesundheit, München-Nürnberg, 29 February–1 March.
5. Crump, K. (1984) A new method for determining allowable daily intakes. *Fund. Appl. Toxicol.* **4**, 854.
6. Allen, B.C., Kavlock, R.J., Kimmel, C.A. and Faustman, E.M. (1994) Dose–response assessment for developmental toxicity. II. Comparison of generic benchmark dose estimates with no observed adverse effect levels. *Fund. Appl. Toxicol.* **23**, 487–495.
7. Johanson, G. and Filser, J.G. (1993) A physiologically based pharmacokinetic model for butadiene and its metabole butadiene

monoxide in rat and mouse and its significance for risk extrapolation. *Arch. Toxicol.* **93**, 151.
8. Paustenbach, D.J. (1995) Retrospective on US health risk assessment: how others can benefit. *Risk: Health, Safety and Environment* **6**, 283.
9. Tardif, R., Lapare, S., Charest-Tardif, G., Bodeur, J. and Krishnan, K. (1995). Physiologically-based pharmacokinetic modelling of a mixture of toluene and xylene in humans. *Risk Analysis* **15**, 335.
10. USEPA (1996) Proposed guidelines for carcinogen risk assessment. EPA/600/P-92/0D3C. Office of Research and Development, Washington, DC.
11. Cotter-Howells, J. and Thornton, I. (1991) Sources and pathways of environmental lead to children in a Derbyshire mining village. *Environ. Geochem. Health* **13**, 127–135.
12. Beck, A.J., Wilson, S.C., Alcock, R.E. and Jones, K.C. (1995) Kinetic constraints on the loss of organic chemicals from contaminated soils: implications for soil-quality limits. *Critical Reviews in Environmental Science and Technology* **25**, 1–43.
13. Mahaffey, K.R. (1977) Quantities of lead producing health effects in humans: sources and bioavailability. *Environ. Health Perspect.* **19**, 285.
14. Vogt, H. (1992) Einfluß von Bentonit and Kieselgur in Legehennanfutler. *Landb. Volkenrode* **42**, 89.
15. Sheppard, S.C. and Evenden, W.G. (1992) Concentration enrichment of sparingly soluble contaminants (V, Th and Pb) by erosion and by soil adhesion to plants and skin. *Environ. Geochem. Health* **14**, 121.
16. WHO (1993) *Guidelines for Drinking-Water Quality*, 2nd edn, vol. 1, *Recommendations*. World Health Organization, Geneva.
17. USEPA (1989) *Risk Assessment Guidance for Superfund*, vol. 1, *Human Health Evaluation Manual* (part A). EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington, DC.
18. HSE (1995) *Occupational Exposure Limits 1995*, EH40/95. HSE Books, Sudbury.

Chapter 4

Receptors: ecosystem health

4.1 Introduction

European concern about environmental issues in the 1960s and 1970s focused on aquatic ecosystems, where some severe problems had become apparent. During the 1980s and in the current decade it has become increasingly accepted that there are also problems associated with terrestrial ecosystems. The identification of ever-increasing numbers of contaminated sites in industrialised countries has emphasised the need for ecological risk assessment. This has resulted in the development of relevant soil tests and other R&D initiatives. Terrestrial risk assessment has a relatively short history, therefore the inclusion of ecological risk assessment (ERA) in general risk assessment of contaminated sites in Europe is fairly recent in origin.[1,2] However, with the growing realisation that sustainable use of soil is vital, more effort is being put into the development of different tools for assessing the risk that contamination poses to terrestrial ecosystems.

The basis of ERA is an exposure assessment and an effects assessment, which are of equal importance. Different test strategies or quality objectives can be used in ERA depending on the level of protection and acceptable risk, e.g. negligible or serious ecological risk. If, however, a substantial level of protection is specified, e.g. 95% or more of the species within an ecosystem, information about the toxicity to the microbial community, plants and soil fauna is usually required. As Cairns[3] stated, the concept of a single most sensitive species is a myth. No single ecotoxicity test, therefore, is sufficient to assess the risk from pollutants to an entire ecosystem. This chapter reviews tests concerning all three groups of organisms.

The Netherlands has developed and used environmental quality objectives to evaluate environmental risk from soil contamination[2] since the early 1990s. Today most countries either already take ERA into

account or are planning to do so in their national risk assessment approaches. The fact that ERA is not formally implemented in many countries does not necessarily imply that it is not applied; there may be cases where the application of ERA is recommended but there is no formal procedural guidance. In some countries where ERA is mandatory it applies to all investigated sites; in others it is mandatory only for some landuse classes (e.g. nature areas) or for certain aspects of a site such as contaminated topsoil. The purpose of conducting environmental risk assessments at contaminated sites may be to ensure sustainable landuse by protecting ecological structure and functions, or more simply to ensure that grass and plants are able to grow in gardens. Some biological assays serve primarily as screening for the presence of hazardous chemicals in the soil, e.g. carcinogenic or mutagenic substances, rather than as an assessment of ecological risk. However, it is important to stress that no matter which methods are used or how ERA is implemented, it is still secondary compared to human risk assessment in all countries.

Although approaches differ at a national or even a regional level, they normally consist of a combination of three general approaches:

1. A comparison of chemical data with generic guideline values or quality criteria derived from toxicity data obtained in standardised ecotoxicological tests (see also Chapter 8). This is widely used in many countries.
2. Bioassays with solid material or soil extracts from the contaminated site as a supplement to chemical analysis. This is common practice in some countries.
3. Monitoring of biomarkers, bioconcentration, indicator species, changes in community structure etc. These indicators are seldom used in current risk assessment practice, but may be employed in research projects.

Many countries consider ERA when setting soil quality objectives, in general environmental policy, and in the development of screening or guideline values used in contaminated site assessments. In some of these countries, soil quality objectives are based on expert judgement, while others use standard toxicity tests and sometimes models. In few cases is bioavailability taken into account. The application of bioassays and biomonitoring is also rare. Although some countries have indicated an intention to increase their use of bioassays and biomonitoring in the future, few have recommended test batteries or developed frameworks for the use of these methods. In the majority of European countries some action is taken when serious ecological risk is identified. The

actions may include remediation, restrictions on landuse, demand for a more detailed site-specific risk assessment or actions to prevent dispersion. Sometimes a combination of actions is applied.

From the information gathered about ERA in different European countries it is obvious that ERA usually consists of rather simplified screening level approaches. Except for the use of soil screening values, there is currently no internationally accepted approach to ERA. However, some commonly used or promising approaches are described in the following sections. The results of current and future R&D programmes will be one of the factors determining how useful these approaches are, and whether or not they will be implemented at a national or international level.

4.2 Ecological screening and guideline values

General approach

The development of remediation programmes in participating countries has led to the generation of a large variety of soil screening values (SSVs). The generic nature of these values makes their application straightforward; the advantages and disadvantages of their use are discussed in greater detail in Chapters 2 and 8. Different countries have different approaches for deriving SSVs, but there are two basic methodologies:

1. Applying a safety factor to reviewed toxicity data. Minor variations exist in the data requirements for different ERA strategies (EU, CSTE, USEPA etc.). However, where few data are available the application of a safety factor of 1000 to the lowest acute LC50 value, or a factor of 100 to the lowest chronic NOEC (No Observed Effect Concentration) value, is generally recommended. If the dataset on chronic effects is sufficiently large and representative, a safety factor of 10 or even 1 (in cases where long-term field data exist) may be applied to the lowest NOEC value. The latter situation is very uncommon in terrestrial risk assessment.
2. Using a statistical extrapolation of toxicity data to derive a PNEC (Predicted No Effect Concentration). Several versions of this method exist,[4–6] their main difference lying in whether they assume the toxicity data to have a log-logistic or a log-normal distribution. The methods are based on statistical analysis of laboratory test data, which takes into account the difference in sensitivity of the test species. Based on a cumulative distribution curve a protection level

of, say, 95% or 50% can be estimated. If the input data to the model are NOEC values, the estimated PNEC will then in principle protect, say, 95% or 50% of the species. The methods assume that the test species collected represent a random sample of species in the ecosystem in question. This may not be true in many cases.

The Potential Affected Fraction (PAF) of species at a fixed soil concentration can be calculated by using sensitivity distribution curves (e.g. those derived by Aldenberg and Slob[4] or Wagner and Løkke[6]). The PAF indicates the percentage of species potentially exposed to concentrations exceeding their NOEC value. Monitoring data can be used to create maps depicting PAFs either for individual chemicals or for the aggregate of all the chemicals in a study.

Derivation of soil screening values

There are several unsolved problems in the derivation of SSVs. The major ones are listed and discussed briefly below:

- Bioavailability
- Risk assessment for mixtures
- Chronic exposure, adaptation and ecological recovery
- Choice of endpoints
- Multiple sources of stress
- Biomagnification
- Soil functioning
- Choice of protection level.

These problems prevent straightforward extrapolation from controlled laboratory experiments to field situations.[7,8] To a certain extent, some of the same problems exist when extrapolating results from site-specific bioassays to field conditions. These major problems are discussed briefly below. More comprehensive discussions are available in the literature.[5,9–11]

Bioavailability

The bioavailability measured in acute laboratory tests of chemicals may be very different from that in the field, due to effects such as ageing. A single generic guideline value cannot allow for the differences in bioavailability, and hence toxicity. Some countries have taken this into account by adjusting the SSV according to the clay and organic matter content of the soil, while others have fixed guideline values for

concentrations in leachates. A clear definition of bioavailability is necessary if it is to be used appropriately in ERA.

The term *bioavailability* covers a wide range of phenomena, including sorption/desorption to soil material and uptake kinetics in soil organisms.[12] Furthermore, metabolism within organisms (e.g. detoxification, storage, excretion) may also influence toxicity. Bioavailability may change with time due to different sorption/desorption processes, behavioural changes, and activation of different detoxification and excretion pathways in exposed organisms.

Risk assessment for mixtures

There is usually more than one pollutant on a contaminated site. However, standardised ecotoxicity tests are often conducted on single chemicals. The toxic response to a chemical mixture may be antagonistic, synergistic or additive, depending on the concentration and mode of action of the chemicals involved.[13,14] It is generally believed that a mixture of chemicals at low concentrations will usually produce additive effects.

Chronic exposure, adaptation and ecological recovery

Ecotoxicity tests normally focus on acute effects, whereas field exposure is likely to be chronic in nature. Several instances of toxic metal adaptation in microorganisms,[15] plants,[16,17] and soil fauna[18,19] have been observed. Pollutant ageing or disappearance from the soil due to leaching, volatilisation or degradation will sometimes enable ecosystems to recover. However, modelling the recovery of ecosystems after disturbance, whether from chemical exposure or otherwise, is a complex process, especially with regard to what constitutes a *normal* ecosystem.[20] All of these aspects make extrapolation from short-term to long-term exposure very complicated.

Choice of endpoints

The data, e.g. NOEC values, used in extrapolations depend on the choice of endpoint; for instance, many experiments only give information about survival, although reproduction and growth are usually far more sensitive parameters. Crommentuijn *et al.*[21] have suggested that the ratio between the lethal effect concentration and the sublethal effect concentration (Sublethal Sensitivity Index, SSI) should be used as an indication of the consequences of chronic stress on life history. The SSI would give some indication of how sublethal function would be maintained under chemical stress, which would be helpful when evaluating the likelihood of pollution-induced effects on populations.

Multiple sources of stress

Laboratory experiments are generally conducted under standardised conditions, to guarantee optimal survival, growth and reproduction of the test species. In nature, however, organisms may be subjected to considerable fluctuation in their environment. Large variations in temperature, humidity, food supply and predator activity may occur throughout the year. Climatic stress may greatly enhance the overall toxicity of a chemical.[22,23] Although this is well documented in the aquatic environment, far less is known about the terrestrial environment. For instance, although van Gestel and van Diepen[24] did not observe any relationship between soil moisture and cadmium toxicity to springtails, the moisture range was within the limits of physiological adaptation. Improving our knowledge about the physiology of test species and of fundamental soil ecology will improve the accuracy of extrapolation from laboratory to field.

Biomagnification

The normal ecological risk assessment procedures for soil do not take into account the potential for biomagnification in terrestrial food chains, e.g. to birds and mammals. However, models that include secondary poisoning or biomagnification in terrestrial food chains have been developed.[25–29] So far all studies indicate that biomagnification is of less concern in terrestrial food-webs than in aquatic ones, and that for most substances the existing SSV may be sufficient to protect larger animals from secondary poisoning.

Soil functioning

The aim of extrapolation methods is to protect soil functions by protecting the structure of the ecosystem (i.e. the species). Some ecotoxicologists and ecologists have questioned this coupling of structure and functioning of ecosystems.[30,31] For example, if species important for the functioning of an ecosystem are among the 5% that a model indicates may be affected, this may impact significantly on the ecosystem as a whole. A large Dutch research programme on functional biodiversity, initiated by RIVM in 1997, is concerned with one of the central questions in ERA: at what level of contamination or stress does the biodiversity (number of species, number of individuals within species, functional groups etc.) decline so far that soil function is endangered to the detriment of the vitality, regeneration and integrity of ecosystems? In the project several indicators for life support functions are being developed.

Choice of protection level

Risk assessors need to consider what kind of ecosystem is to be protected before choosing a protection level. The degree of protection may differ significantly according to location (e.g. urban areas *versus* national parks) or potential future landuse. Even at a high protection level, problems may arise. For example if a rare species of national concern is among the most vulnerable species, this may pose an ethical problem for regulators and society even though effects on the ecosystem itself may not be critical.

Although there is considerable debate over the merits of SSVs, they are central to the ecological risk assessment strategy of most countries. This is because they (i) provide a basis for comparison between sites, (ii) streamline the screening of contaminated sites and (iii) may facilitate communication between regulators and the public or other interested parties.[11,32] Therefore, even in the long term, it will be neither possible nor advisable to replace SSVs completely. SSVs may also serve as limits when calculating critical loads of substances involved in long-range transboundary air pollution.

Ecotoxicological test systems for establishing generic SSVs

One of the principles underpinning the derivation of screening values for contaminated soil is the use of internationally accepted standardised soil test procedures similar to those used in the risk assessment of chemicals. Few exist today, but several tests appear promising and will probably be prepared for standardisation in the near future. The emphasis of these prognostic tests is on reproducibility. Existing standardised ecotoxicity test methods were mainly developed to evaluate the effect of specific test substances, e.g. in risk assessment of new and existing chemicals in the EU (Directive 93/67/EEC and Regulations 793/93 and 1488/94). Most of the standardised tests are specific to the aquatic environment, and use of terrestrial test organisms is still in its infancy. Consequently, the Technical Guidance Document for risk assessment of existing chemicals contains only a provisional strategy for risk to the terrestrial environment, since it is recognised that few experimental data are available.

Toxicity data from non-standardised tests with terrestrial organisms may be found in the published literature. Some of the available tests for ERA of chemicals are reviewed below. A selection of terrestrial soil tests can be found in references 1, 33–37 and 55. In this chapter testing guidelines from OECD and ISO are described together with terrestrial

tests in the CARACAS database. It should also be mentioned that the American Society for Testing and Materials (ASTM) has a programme of testing that includes many valuable guidelines.

One of the major tasks of ecological risk assessment is to evaluate the potential risks to a wide variety of terrestrial organisms from the 100,000 or more chemicals used in the EU. Relationships between the properties of some chemicals and their biological action or environmental fate have been established using Quantitative Structure–Activity Relationships (QSARs).[38–40] If it also proves possible to establish similar relationships for species sensitivity – that is, Quantitative Species Sensitivity Relationships (QSSRs)[41–44] – this may reduce the number of test organisms necessary for risk assessment.

Tests with microorganisms

More than two-thirds of the soil biomass consists of microorganisms. Microbial communities are essential for the decomposition and degradation of complex organic substances, including toxic chemicals. They are indispensable in the natural cycles of carbon, nitrogen, phosphorus and sulphur that are crucial in maintaining a sustainable soil quality for soil fauna and plant growth. Several species of bacteria and fungi may play similar roles in maintaining soil functions. This complicates evaluation of soil contamination because elimination of some species may increase the number of others so that the overall functioning of the soil is unchanged. However, in the long term, this sort of reduction of microfloral variation may make the soil less tolerant of other changes, e.g. of a chemical or climatic nature. For example, microorganisms tolerant to toxic metals may degrade complex organic compounds like PAHs more slowly than species not adapted to elevated toxic metal concentrations, or they may be less tolerant of climatic stress and hence have a higher mortality rate under uncontaminated conditions. This principle is called *cost of tolerance*.[18]

Several endpoints have been used in studying the effects of chemicals on microflora,[45] e.g. soil respiration (CO_2 produced), and enzyme (dehydrogenase, phosphatase and urinase) activity. It is important to choose an endpoint and group of microorganisms that are relatively sensitive to chemical stress. Nitrifying bacteria, which are responsible for the oxidation of ammonium to nitrite and then to nitrate, are generally considered a very sensitive group of organisms, and nitrification is an essential pathway in nitrogen mineralisation. Standard test methods for nitrogen mineralisation are available in the ISO test programme (ISO 14238). Furthermore, some national programmes have set protocols or guidelines for assessing the effects of

chemicals in soil. One example is the Swedish MATS programme, which includes several guidelines for assessing the effects of chemicals on the soil microflora, especially on nitrogen mineralisation processes such as nitrification, denitrification and nitrogen fixation.[46] Standardised test methods for determining soil biomass or soil respiration are also available (ISO 14240), but these are rather insensitive endpoints and should only be used as a general indication of the overall fertility of a soil.

Tests with plants

Plant roots in soil have a very large biologically active surface. There is a wide range of phytotoxicity tests for evaluating the effects of pesticides and other chemicals,[47,48] and a few of these are internationally standardised, e.g. OECD 208 (currently being updated) and ISO 1129. The most common endpoints in plant tests are inhibition of root growth, and inhibition of emergence and growth of higher plants. The recommended test species are generally well-defined crops with a fast growth rate (e.g. bean, cabbage, cress, lettuce, oat, rape, rye grass, wheat, turnip) and do not include species that display significant variation in lifestyle and/or genetic composition. There is a need for the development and validation of one or more chronic tests in soil that would cover the whole plant lifecycle; a recent development using a rapid-cycling brassica species appears to be promising.[49] The species (a variant of wild mustard) undergoes a complete lifecycle from seed to seed in approximately 35 days, so measurements can be conducted on the vegetative and flowering stages as well as on reproductive endpoints, e.g. seed production and emergence. The German project *Biological Methods for Soil Remediation: Ecotoxicological Test Batteries* includes research into developing and standardising a chronic whole-lifecycle plant test, using turnip and oat.

Tests with soil fauna

Soil fauna covers a very wide variety of animals. Due to the great variation in physiology, lifestyle and place in the food web, the nature of exposure may vary considerably. For example, nematodes live mainly in the soil pore water and have little contact with soil particles. In contrast, earthworms may ingest large amounts of soil, thus increasing their chance of taking up contaminants bound to soil particles. Some insects have soft skins whereas others have a hard cuticle that will severely restrict the absorption of soluble chemicals. Some invertebrates never leave the soil matrix, while others feed above ground but breed and hatch in the soil. Another important group of terrestrial invertebrates

lives in the litter or vegetation above the soil surface and is only indirectly affected by soil contamination.

The acute toxicity tests with earthworms prescribed by OECD (207), ISO (11268-1) and EU (TM C.8) are currently the only tests accepted as international standards. These OECD and EU tests include three different standard acute tests, including exposure through an artificial soil (10 % Sphagnum peat, 20% kaolin clay and 70% industrial quartz sand). Sorption data can be used to extrapolate the results from the artificial soil to natural soils.

Although these tests recommend the use of the species *Eisenia foetida* and *E. andrei* for practical reasons, the selection of test species is not restricted to these two. Sheppard and Evenden[50] tried to modify the design of the earthworm survival tests in soil to facilitate the use of, for example, *Lumbricus* species. On the basis of a comparison with the toxicity of 23 pesticides to *Lumbricus terrestris*, Heimbach[51] concluded that *E. foetida* could be used as a representative species for testing the toxicity of chemicals to earthworms. On the basis of a large literature review, Edwards and Coulson[52] also concluded that the compost worm was suitable for an initial screening of chemicals. According to this review, application of a safety factor of 10 would bring *E. foetida* in line with the most sensitive species, normally *Apporectodea caliginosa*.

ISO is currently preparing protocols for determining reproductive effects on *E. foetida* (ISO 11268-2-draft) and on the springtail *Folsomia candida* (ISO 11267-draft). The German institute ETC Ökotoxikologie GmbH is chairing an interlaboratory comparison for validating the OECD Draft Guideline *Enchytraidae Reproduction Test*.[53] Some other ecotoxicity tests for the effects of contaminated soils on soil invertebrates are in regular use.[54,55] In 1997 the EU research project SECOFASE[56] was completed. The main aim of this international project was the development, improvement and standardisation of tests for assessing sublethal effects of chemicals on soil fauna. The project was divided into 10 sub-projects, each concerned with an individual taxon but using common principles for test conditions and endpoints. The taxonomic groups involved were Nematoda, Enchytraeidae, Lumbricidae, Collembola, Oribatida, Gamasida, Staphylinidae, Isopoda, Diploda and Chilopoda. In total, 17 draft protocols were prepared in a standardised format suitable for international adaptation as draft test guidelines. Of these, four are ready for international inter-calibration. These are: two sublethal toxicity tests with the collembolan species *Folsomia fimetaria* and *F. candida*, a sublethal toxicity test with the earthworm *E. foetida* and a sublethal two-species toxicity test with the gamasid mite *Hypoaspis aculeifer*, which preys on the collembola *F. fimetaria*.

4.3 Biological assays

Introduction

Ecotoxicity tests are usually performed in soil (often a standard soil) that has been spiked with the chemicals in a concentration range suitable for estimating LC50, EC50 or NOEC values. These tests are mostly used for generic hazard or risk assessment. In contrast, biological tests (bioassays) use soil from the contaminated site in question and hence can be used to produce a more site-specific risk assessment. Many of the terrestrial ecotoxicity tests reviewed above were originally designed to determine the effect of a substance *added* to the soil. Therefore, these tests cannot be used to calculate *effect* or *no effect* values for field-contaminated soil in the same manner. If there are only a few chemicals and there is a range of concentrations, it may be possible to estimate EC50 or NOEC values. Otherwise, the reaction of the test organisms must be compared with an uncontaminated control before estimating the lowest dilution factor necessary to produce a fixed effect, e.g. the level of the control or 10% or 50% below the control. It will often be very difficult to find a control soil resembling the contaminated material in its chemico-physical parameters, nutrient content and microbial activity.

Bioassays do not often form part of the risk assessment of contaminated sites. Their use has been mainly restricted to research projects. However, their use in national ERA strategies is increasingly seen as a useful supplement to chemical analysis in decisions related to contaminated land. Although bioassays are generally versatile, organisms from different levels of biological organisation should be included in a test battery of assays to improve reliability and protect the ecosystem. Whereas cost limits the number of possible pollutants that can be investigated by conventional chemical analysis, bioassays can detect the effects of a wide range of toxins at relatively low cost. Moreover, these assays will be able to monitor increases in toxicity caused by synergistic effects of contaminants, and metabolites formed during degradation. A matrix-specific change in bioavailability will occur over time, due to sorption and sequestration of contaminants. This may not be detectable by standard chemical analysis and is rarely considered when deriving soil quality objectives.

For heavily contaminated sites evaluating different soil quality parameters is of less importance. In these cases the primary concern is usually the possibility of toxic substances dispersing to surrounding ecosystems, e.g. groundwater and surface waters, causing adverse

effects. Tests with extracts or pore water have been used in many situations as a relatively fast and easy method of testing the bioavailability and toxicity of chemicals in solid material. Different leaching solutions can be used to extract the contaminants from the solid material while the pore water can be collected by centrifuging. Each extract solution may be representative of a different degree of bioavailability and dispersal risk. In the overall context of risk assessment, screening tests with soil extracts may be very useful for hazard identification and for more refined exposure assessment. A considerable number of bioassays using leachates of contaminated soils or polluted solids are now available and are regularly used on their own or as support for contact tests.[57,58] When using ecotoxicity tests with extracts to predict effects in soil, relationships between data obtained with extracts in the laboratory must be validated against effects observed in the contaminated material because the assays may be sensitive to other parameters such as conductivity or ammonium. The choice of assay must be made according to the purpose of the investigation. For examining possible groundwater contamination or identifying potential risks to surface waters, bioassays using leachates are the natural choice. For a comprehensive ERA of contaminated sites both leachate tests and contact tests with solids may be required.

Bioassays using microorganisms

As it is currently very difficult and time consuming to measure the species composition and structure of a microbial community using a dilution plate technique, other methods have recently been introduced. These include RNA or DNA analysis, and analysis of the phospholipid fatty acids composition.[59] Although such methods may provide some information about species diversity, the results cannot be directly related to soil functions. There have been some interesting recent developments in the field of microbial testing. The BIOLOG plate system consists of a number of small cells, each containing a well-defined carbon source, and a control without any carbohydrate source. The growth of bacteria within each cell is measured by the release of a violet end-product, resulting from the reaction of the respiration product with tetrazolium violet. This method makes it possible to test the effect of pollutants on many different processes simultaneously. Solid-phase bioassays involve direct contact between solid material and the bacterium. Microtox solid-phase uses the marine bacterium *Vibrio fisheri*, whereas other bioassays use soil bacteria such as *Bacillus cereus*, *Alcaligenes eutrophus* or *Pseudomonas fluorescens*.[60] The

advantage of these test systems is that the effect of the soil matrix on bioavailability is taken into account. A test using *Rhizobium meliloti*, which forms a symbiotic association with the root of a legume, also involves direct contact between the matrix and the bacteria because the contaminated soil is mixed with mineral agar prior to testing.[61] The endpoints are nodulation and plant growth.

Microbial bioassays can also be used for liquid samples, either for leachates sampled in the field as in the case of landfills,[62] or for extracts obtained by leachate testing of solid material in the laboratory. These bioassays are the same as those for waste water and effluents, the most common ones being the standardised tests like Microtox, Toxi-Chromotest (*Escherichia coli*), MetPad (*E. coli*) and other luminescent bacteria like *Photorhabdus luminescens*. These rapid tests can also be very useful in a toxicity identification evaluation approach. Ecological realism in soil extract testing can be improved by using protozoans[63] belonging to different trophic levels.

Bioassays with plants

Bioassays with plants are useful for estimating adverse effects not easily detected by chemical analysis. The standard tests described above are usually also applicable to site-specific assessments. There may, however, be considerable species variation because the storage and degradative capacities of plants differ according to the species and pollutant.

Plant testing for algae and higher plants with soil extracts is also in regular use; examples include monitoring for seed germination, root elongation of crops and the number of fronds of the common duckweed *Lemna minor*.[48,64,65] The disadvantage of algae (*Pseudokirchneriella, Scenedesmus* or *Chlorella* spp.) for some applications is that they only tolerate low salt concentrations. However, in a test on leachates from salt mines, a *Chlamydomonas* (green algae) species displayed a high salt tolerance.[66]

Bioassays with soil fauna

An ecotoxicological evaluation of the impact of contamination on soil fauna should include testing or monitoring of several taxonomically different groups. A number of more or less standardised soil tests using invertebrates are also appropriate for site-specific assessments (see above). The testing of leachates using invertebrates is common in

terrestrial risk assessment. This is because these tests are cost effective and simple to use. They are also relevant for estimating potential ecological effects of soil contaminants on groundwater and freshwater systems. Tests include those using daphnia (OECD 202, EU C.2), nematodes[67–69] and enchytraeidae.[70]

4.4 Potential approaches for future application

Ecological risk assessment of contaminated sites is of fairly recent origin. There are several difficulties associated with assessing the risk posed by chemicals to natural ecosystems that cannot readily be taken into account using current methods. Natural ecosystems are very complex and exhibit a variety of responses to natural and anthropogenic disturbances. Community dynamics and organisation are too complicated to allow simple risk assessment based on acute laboratory experiments.[71] Changing the number of species or eliminating others may lead to unanticipated responses at community level. Therefore, any attempt to extrapolate from lower to higher levels of organisation must involve careful consideration of factors such as the number of trophic levels, the exposure history, and the frequency and duration of other disturbances.[71] Multispecies bioassays or whole community analysis could represent a significant improvement in current risk assessment methods.

Improving the risk assessment of ecosystems will involve refining current methods as well as designing new ones. Both will depend on the outcome of current and future R&D projects, especially with regard to establishing proper validation procedures. The following approaches appear to be promising:

- Biomonitoring
- Biochemical markers
- Multispecies assays
- Pollution-induced community tolerance
- Community structure analysis

Biomonitoring

Biological monitoring may be a useful tool to supplement more traditional ERA based on guideline values and bioassays. The simplest form of biomonitoring is measuring the concentration of one or more chemicals in the tissue of selected invertebrates or plants, e.g. toxic metals in earthworms, snails or plants. The contaminant concentration

in animals or plants may be more useful for exposure assessment than total soil concentration alone because it provides information about bioavailability and bioconcentration. However, due to significant differences in tolerance, adaptation, detoxification mechanisms etc. the tissue concentration does not in itself give precise information about the ecological risk caused by a pollutant. Consequently, attempts have been made to establish critical body concentrations or internal threshold concentrations for pollutants in soil invertebrates and plants.[72–74] Use of the latter, together with information about field concentration factors, may result in better estimates of critical soil concentrations than laboratory experiments. Several problems must be addressed before body concentration factors (BCFs) can be considered sufficiently developed to be applicable to the derivation of soil quality criteria. For example BCF tends to decrease with increasing soil concentration, and for some essential metals like zinc and copper the internal body level is strictly regulated. Furthermore, the general state of health of an organism, and conditions such as soil pH and food supply, may affect the critical body concentration. It is therefore essential to test whether internal concentrations are the same in the field as in the laboratory.

The locomotive behaviour of certain soil invertebrates, e.g. woodlice, has been used successfully as an indicator of soil contamination. Aspects of locomotive and other behaviour, e.g. pathlength and movement velocity, are generally good indicators of chemical stress and can be analysed using linear discriminant analysis. If such discriminant models have a sufficiently solid foundation, they have considerable predictive value. Analysis has shown that control populations of woodlice were remarkably similar in their locomotive behaviour despite a geographic separation of more than 300 km, and that the locomotive behaviour of animals from polluted sites was significantly different from that of controls.[75,76]

Biochemical markers

Biomarkers are substances that produce a 'biological response that can be related to an exposure to, or toxic effect of chemicals'.[77] A large number of biochemical/cellular markers has been suggested for ERA.[78] The major reason behind the current interest in biomarkers is their potential to circumvent the limitations of classical approaches.[79] The question of chemical bioavailability is overcome because biomarkers only quantify biologically available pollutants. This is especially relevant for soil because bioavailability varies widely with different soil characteristics. Also, biomarkers can be used in both laboratory and field conditions even when there is a complex mixture of pollutants.

Biochemical and cellular markers have the advantage of rapid response times, and may be specific to particular classes of chemical.[80] However, despite their applicability to laboratory and field conditions, their ecological significance is often questionable. This is because biochemical changes indicate exposure (and the resultant induction of protective or homeostatic mechanisms) rather than damage. Links between the induction of a biomarker and effects on higher levels of organisation have been established only for a few biomarkers. Thus, the most important role of biomarkers in ecotoxicological research has so far been to clarify mechanistic links between the molecular, biochemical and physiological levels. However, there are some promising applications of biomarkers in ERA, for instance the use of lysosomal membrane stability in earthworms as a marker for soil copper pollution.[81,82] Biomarkers may also be useful tools for monitoring genotoxic compounds. For example, earthworms may be suitable organisms for assessing accumulated risk from toxic waste because of the nature of their exposure and feeding patterns. The Alkaline Comet Assay, for example, will measure the cumulative DNA damage caused by all genotoxic pollutants available to the organism.[83]

Multispecies assays

Although single-species laboratory assays may be effective for examining the relative toxicity of chemical substances, for a number of reasons they may not be the most accurate or efficient method of predicting responses in contaminated ecosystems.[7,84] They do not take species interaction into account and they are often conducted under physical and chemical conditions that do not resemble natural habitats. Therefore many workers in this field have recommended the use of multispecies test systems, the so-called microcosms or mesocosms. Soil mesocosms (or microcosms) have been defined in the literature as experimental units in which the response of more than one biotic species is measured and where at least one of the biotic species is at a higher trophic level than microorganisms. Furthermore, the assay should be long term because indirect changes caused by species interactions will occur over a longer timescale than direct acute effects.

Although it is unlikely that multispecies assays in microcosms and mesocosms will become as standard as single-species tests, studies have documented their usefulness in predicting the impact of chemicals on ecosystems.[85,86] Furthermore, observations of how species interact in the presence of toxins can be used in the development of predictive computer models.[87] Mesocosms are therefore valuable tools for predicting whole ecosystem responses to stress and may come into their own in

future ecological risk assessment. Several workers have recommended that ecological risk assessment should be at least partly based on the use of microcosm and mesocosm studies.[71,88,89] Van Straalen and Løkke[90] argue that due to their complexity, lack of transparency and large variations in outcome, multispecies tests are *black boxes* that do not give insight into the interactions between different ecosystem components. Hence they regard them as unsuitable for regulatory purposes on their own.

Pollution-induced community tolerance

The pollution-induced community tolerance (PICT) concept[91] is a very promising approach for analysing the effects of pollutants on aquatic algae communities. By comparing the sensitivity of algal communities along a concentration gradient, it is possible to estimate the level of pollution at which the community has developed tolerance towards the pollutants, and hence is under detectable chemical stress. A few studies indicate that PICT can also be used in the terrestrial environment. Díaz-Raviña *et al.* observed increased tolerance in microflora communities that were pre-exposed for nine months to toxic metals added to the soil.[92] They observed increased community tolerance for Cu, Cd, Zn and Ni but not for Pb. Furthermore, they found that tolerance to one toxic metal often led to increased tolerance for other metals. This and other examples show that the PICT concept can be used for microorganisms in the terrestrial environment. Furthermore, as soil sampling along a concentration gradient is practicable at many sites, this approach may make it possible to identify soil concentrations at which no community tolerance has evolved. Although not fully validated, Rutgers *et al.*[93] have successfully used PICT in combination with the BIOLOG plate system as a rapid and robust risk assessment method for contaminated soils. They used BIOLOG to construct 80–90 different concentration–response relationships for field microbial communities along a contaminant gradient (zinc) in the field. The results showed that PICT had become established below the Dutch intervention value for zinc.

Structure analysis of whole communities

A well-designed and replicated terrestrial study using mesocosms is often time consuming and costly. Furthermore, the results may not be directly applicable because mesocosms may not give sufficient insight into the interactions between the different components of an ecosystem. Therefore, on-site monitoring of changes in community structure may

provide a low-cost predictive tool for assessing effects at a high level of biological organisation.

Whole-community analysis to monitor the ecological impact of pollutants on flora and fauna at specific contaminated sites has been used successfully in a few cases.[94–97] Multivariate statistical analysis of communities may help in deciding whether changes to an ecosystem are significant and what constitutes an unacceptable level of disturbance. Although community structure analyses seem promising, because they are rapid and relatively cheap, they still have some limitations. For an assessment to be relevant to the whole ecosystem, the distribution of organisms that are sensitive to the chemicals present at a site must be mapped. This is because there are no safety factors inherent in this method. Mapping may be performed by monitoring a wide range of different species and by knowing the principal pollutants, for instance on the basis of historical records. Finally, as in the case of bioassays, it is necessary to find a control site which resembles the affected area in all or most of the abiotic parameters that influence the health of plants and animals, e.g. organic matter, water content, pH and compaction. For retrospective ecological risk assessments, in which the objective is to measure the ecological effects of historical contamination, the identification of species diversity and abundance over time and space seems to offer a logical, straightforward and unambiguous approach.

4.5 Concluding remarks

This chapter has outlined some of the most common risk assessment methods for terrestrial ecosystems. Generally, most countries use generic guidelines for a first screening of ecological risk. Where further assessment is necessary a range of bioassays may be used to assess the site-specific ecological impact or to rank a number of sites according to ecological risk. The advantages and disadvantages of these regularly used methods have been discussed and it is evident that there is a need for further development, for instance by introducing new test species and/or more sensitive endpoints in standardised tests. Most of all, further validation is required in order for the results of controlled single-species laboratory experiments to be extrapolated to *real-world* situations in the field. It is very important to keep the objectives of ERA in mind. As with human health risk assessment, a major objective is often to determine whether or not remediation is necessary and, where required, to help specify appropriate remediation targets.

References

1. Sheppard, S.C., Gaudet, C., Sheppard, M.I., Cureton, P.M. and Wong, M.P. (1992) The development of assessment and remediation guidelines for contaminated soils, a review of the science. *Canadian Journal of Soil Science* **72**, 359–394.
2. van Leeuwen, K. (1990) Ecotoxicological effects assessment in the Netherlands: recent developments. *Environmental Management* **14**, 779–792.
3. Cairns, J. (1986) What is meant by validation of predictions based on laboratory toxicity tests? *Hydrobiologia* **137**, 271–278.
4. Aldenberg, T. and Slob, W. (1993) Confidence limits for hazardous concentrations based on logistically distributed NOEC data. *Ecotoxicology and Environmental Safety* **25**, 48–63.
5. van Straalen, N. and Denneman, C.A.J. (1989) Ecotoxicological evaluation of soil quality criteria. *Ecotoxicology and Environmental Safety* **18**, 241–251.
6. Wagner C. and Løkke H. (1991) Estimation of ecotoxicological protection levels from NOEC toxicity data. *Water Research* **25**, 1237–1242.
7. Spurgeon, D.J. (1997) Extrapolation of laboratory toxicity results to the field: a case study using the OECD artificial soil earthworm test. In: *Ecological Risk Assessment of Contaminants in Soil* (N.M. van Straalen and H. Løkke, eds). Ecotoxicology Series vol. 5. Chapman & Hall, London, pp. 253–274.
8. van Beelen, P. and Notenboom, J. (1996) Validation of toxicity data and risk levels for soil: progress report 1995. RIVM Report no. 607 505 002. The Netherlands.
9. Løkke, H. (1994). Ecotoxicological extrapolation: tool or toy? In: *Ecotoxicology of Soil Organisms* (M. Donker, H. Eijsackers and F. Heimbach, eds). CRC Press, Boca Raton, FL, pp. 411–425.
10. van Gestel, C.A.M. (1997) Scientific basis for extrapolating results from soil ecotoxicity tests to field conditions and the use of bioassays. In: *Ecological Risk Assessment of Contaminants in Soil*, (N.M. van Straalen and H. Løkke, eds). Ecotoxicology Series vol. 5. Chapman & Hall, London, pp. 25–50.
11. van Straalen, N.M. (1993) Open problems in the derivation of soil quality criteria from ecotoxicity experiments. In: *Contaminated Soil '93. Fourth International Conference on Contaminated Soil.* (F. Arendt, G.J. Annokkee, R. Bosman and W.J. van den Brink, eds). Kluwer, Dordrecht, pp. 315–326.

12. Peijnenburg, W.J.G.M., Posthuma, L., Eijsackers, H.J.P. and Allen, H.E. (1997) A conceptual framework for implementing of bioavailability of metals for environmental management purposes. *Ecotoxicology and Environmental Safety* **37**, 163–172.
13. Posthuma, L., Weltje, L. and Anton-Sanchez, F.A. (1996) Joint toxic effects of cadmium and pyrene on reproduction and growth of the earthworm *Eisenia andrei*. RIVM report no. 607 506 001, pp. 1–38. The Netherlands.
14. van Gestel, C.A.M. and Hensbergen, P.J. (1997) Interaction of Cd and Zn toxicity for *Folsomia candida* in relation to bioavailability. *Environmental Toxicology and Chemistry* **16**, 1177–1186.
15. Trevors, J.T., Oddie, K.M. and Belliveau, B.H. (1985) Metal resistance in bacteria. *FEMS Microbiological Reviews* **32**, 39–54.
16. Ernst, W.H.O., Verkleij, J.A.C. and Schat, H. (1992) Metal tolerance in plants. *Acta Botanica Neerl.* **41**, 229–248.
17. Macnair, M.R. (1987) Heavy metal tolerance in plants: a model evolutionary system. *Trends in Ecology and Evolution* **2**, 354–359.
18. Posthuma, L. and van Straalen, N.M. (1993). Heavy metal adaptation in terrestrial invertebrates: a review of occurrence, genetics, physiology and ecological consequences. *Comparative Biochemistry and Physiology* **106C**, 11–38.
19. van Straalen, N.M. and Donker, M.H. (1994) Heavy metal adaptation in terrestrial arthropods: physiological and genetic aspects. *Protocol of Experimental and Applied Entomology* **5**, 3–17.
20. Bender, E.A., Case, T.J. and Gilpin, M.E. (1984). Perturbation experiments in community ecology: theory and practice. *Ecology* **65**, 1–13.
21. Crommentuijn, T., Doodeman, C.J.A.M., van der Pol, J.C.C., Doornekamp, A., Rademaker, M.C.J. and van Gestel, C.A.M. (1995). Sublethal sensitivity index as an ecotoxicity parameter measuring energy allocation under toxicant stress: application to cadmium in soil arthropods. *Ecotoxicology and Environmental Safety* **31**, 192–200.
22. Holmstrup, M. (1997) Drought tolerance in *Folsomia candida* Willem (Collembola) after exposure to sublethal concentrations of three soil-polluting chemicals. *Pedobiologia* **41**, 354–361.
23. Zachariassen, K.E. and Lundheim, R. (1995) Effects of environmental pollutants on the cold-hardiness of arctic and boreal ectothermic animals. In: *The Contaminants in the Nordic Ecosystem: Dynamics, Processes and Fate* (M. Munawar and M. Luotola, eds). SPB Academic Publishing, Amsterdam.

24. van Gestel, C.A.M. and van Diepen, A.M.F. (1997) The influence of soil moisture content on the bioavailability and toxicity of cadmium for *Folsomia candida*. *Ecotoxicology and Environmental Safety* **36**, 123–132.
25. Goree, M., Tamis, W.L.M., Traas, Th.P. and Elbers, M.A. (1995) BIOMAG: a model for biomagnification in terrestrial food chains. The case of cadmium in the Kempen, The Netherlands. *The Science of the Total Environment* **168**, 215–223.
26. Jongbloed, R.H., Peijnenburg, J., Mensik, B.J.W.G., Traas, Th.P. and Luttik, R. (1994) A model for environmental risk assessment and standard setting based on biomagnification. Top predators in terrestrial ecosystems. RIVM report no. 719 101 012. The Netherlands.
27. Luttik, R., Traas, Th.P. and de Greed, J. (1992) Incorporation of biomagnification in procedures for environmental risk assessment and standard setting. RIVM report no. 719 101 005. The Netherlands.
28. Romijn, C.A.F.M., Luttik, R. and Canton, J.H. (1994) Presentation of a general algorithm to include effect assessment on secondary poisoning in the derivation of environmental quality criteria. *Ecotoxicology and Environmental Safety* **24**, 107–127.
29. Spurgeon, D.J. and Hopkin, S.P. (1996) Risk assessment of the threat of secondary poisoning by metals to predators of earthworms in the vicinity of a primary smelting works. *The Science of the Total Environment* **187**, 167–183.
30. Forbes, T.L. and Forbes, V.E. (1993) A critique of the use of distributed-based extrapolation models in ecotoxicology. *Functional Ecology* **7**, 249–254.
31. Smith, E.P. and Cairns, J. (1993) Extrapolation methods for setting ecological standards for water quality: statistical and ecological concerns. *Ecotoxicology* **2**, 203–219.
32. Vegter, J.J., Roels, J.M. and Bavinck, H.P. (1988) Soil quality standards: science or science fiction. In: *Contaminated Soil '88.* (K. Wolf, W.J. van den Brink and F.J. Colon, eds). pp. 309–316.
33. Canadian Council of Ministers of the Environment (1997) *A Framework for Ecological Risk Assessment: Technical Appendices.* Canadian Council of Ministers of the Environment, Canada.
34. DECHEMA (1995) *Bioassays for Soil. Methods for Toxicological/Ecotoxicological Assessment of Soils* (Kreysa and Weisner, eds). Frankfurt.
35. Keddy, C., Greene, J.C. and Bonnell, M.A. (1994) *A Review of Whole Organism Bioassays for Assessing the Quality of Soil, Freshwater*

Sediment and Freshwater in Canada. Environment Canada, Scientific Series no. 198. Ottawa, Ontario, 183 pp.
36. Samsø-Pedersen, L. and Pedersen, F. (1994) Discussion paper regarding terrestrial effect assessment. Final Draft. Prepared for the Organization for Economic Cooperation and Development (OECD). VKI, Water Quality Institute, Denmark, 64 pp.
37. Tarradellas, J., Bitton, G. and Rossel, D. (1997) *Soil Ecotoxicology*. CRC Lewis Publishers, Boca Raton, FL, 286 pp.
38. Paasicirta, J., Paukku, R. and Sinkkonen, S. (1993) Structural organic chemistry in ecotoxicology. *Chemosphere* **27**, 445–460.
39. Sabljic, A. (1991) Chemical topology and ecotoxicology. *The Science of the Total Environment* **109**, 197–220.
40. van Gestel, C.A.M., Ma, W. and Els Smit, C. (1991) Development of QSARs in terrestrial ecotoxicology: earthworm toxicity and soil sorption of chlorophenols, chlorobenzenes and dichloroaniline. *The Science of the Total Environment* **109**, 589–604.
41. Blanck, H. (1984) Species dependent variation among aquatic organisms sensitivity to chemicals. *Ecological Bulletins* **36**, 107–119.
42. Eijsackers, H. and Løkke, H. (1996) Soil ecotoxicological risk assessment. *Ecosystem Health* **2**, 259–270.
43. Kammenga, J.E., van Gestel, C.A.M. and Bakker, J. (1994) Patterns of sensitivity to cadmium and pentachlorophenol among nematode species from different taxonomic and ecological groups. *Archives of Environmental Contamination and Toxicology* **27**, 88–94.
44. Kooyman, S.A.L.M. (1987) A safety factor for LC50 values allowing for differences in sensitivity among species. *Water Research* **21**, 269–276.
45. Torstensen, L. (1997) Microbial assays in soils. In: *Soil Ecotoxicology* (J. Tarradellas, G. Bitton and D. Rossel, eds). CRC Lewis, Boca Raton, FL, pp. 207–234.
46. Torstensson, L. (ed.) (1993) *Guidelines. Soil Biological Variables in Environmental Hazard Assessment (MATS)*. Swedish Environmental Protection Agency.
47. Verkleij, J.A.C. (1994) Effects of heavy metals, organic substances and pesticides on higher plants. In: *Ecotoxicology of Soil Organisms* (M. Donker, H. Eijsackers and F. Heimbach, eds). CRC Lewis, Boca Raton, FL, pp. 139–161.
48. Wang, W. (1991) Literature review on higher plants for toxicity testing. *Water, Air and Soil Pollution* **59**, 381–400.
49. Sheppard, S.C., Evenden, W.G., Abboud, S.A. and Stephenson, M. (1993) A plant life-cycle bioassay for contaminated soil, with

comparison to other bioassays: mercury and zinc. *Archives of Environmental Contamination and Toxicology* **25**, 27–35.
50. Sheppard, S.C. and Evenden, W.G. (1992) Optimised design for earthworm survival tests in soil. *Bulletin of Environmental Contamination and Toxicology* **49**, 648–655.
51. Heimbach, F. (1985). Comparison of laboratory methods, using *Eisenia foetida* and *Lumbricus terrestris*, for the assessment of the hazard of chemicals to earthworms. *Zeitschrift für Pflanzenkrankheiten und Pflanzenschutz* **92**, 186–193.
52. Edwards, P.J and Coulson, J.M. (1992) Choice of earthworm species for laboratory tests. In: *Ecotoxicology of Earthworms* (P.W. Greig-Smith, H. Becker, P.J. Edwards and F. Heimbach, eds). Intercept Ltd, Andover, pp. 36–43.
53. Römbke, J. (1989) *Enchytrateus albidus* as a test organism in terrestrial laboratory systems. *Archives of Toxicology* **Suppl. 13**, 402–405.
54. Reinecke, A.J. (1992) A review of ecotoxicological test methods using earthworms species. In: *Ecotoxicology of Earthworms.* (P.W. Greig-Smith, H. Becker, P.J. Edwards and F. Heimbach, eds). Intercept Ltd, Andover, pp. 7–19.
55. van Gestel, C.A.M. and van Straalen, N.M. (1994) Ecotoxicological test systems for terrestrial invertebrates. In: *Ecotoxicology of Soil Organisms* (M. Donker, H. Eijsackers and F. Heimbach, eds). CRC Lewis, Boca Raton, FL, pp. 205–228.
56. Løkke, H. and van Gestel, C.A.M. (eds) (1998) *Handbook of Soil Invertebrate Toxicity Tests.* John Wiley and Sons, Chichester.
57. Debus, R. and Hund, K. (1997). Development of analytical methods for the assessment of ecotoxicological relevant soil contamination. Part 2: Ecotoxicological analysis in soil and soil extracts. *Chemosphere* **35**, 239–261.
58. Wahle, U. and Kördel, W. (1997) Development of analytical methods for the assessment of ecotoxicological relevant soil contamination. Part 1: Development and improvement of soil extraction methods for the determination of the bioavailable parts of contaminants. *Chemosphere* **35**, 223–237.
59. Pennanen, T., Frostegard, A., Fritze, H. and Baath, E. (1996) Phospholipid fatty-acid composition and heavy-metal tolerance of soil microbial communities along 2 heavy metal-polluted gradients in coniferous forests. *Applied and Environmental Microbiology* **62**, 420–428.

60. Rönnpagel, K., Liß, W. and Ahlf, W. (1995) Microbial bioassays to assess the toxicity of solid-associated contaminants. *Ecotoxicology and Environmental Safety* **31**, 99–103.
61. Wetzel A. and Werner D. (1995) Ecotoxicological evaluation of contaminated soil using the legume root nodule symbiosis as effect parameter. *Environmental Toxicology and Water Quality* **10**, 127–133.
62. Clément, B., Persoone, G., Janssen, C. and Le Dû-Delepierre, A. (1996). Estimation of the hazard of landfills through toxicity testing of leachates. *Chemosphere* **33**, 2303–2320.
63. Forge, T.A., Berow, M.L., Darbyshire, J.F. and Warren, A. (1993) Protozoan bioassays of soil amended with sewage sludge and heavy metals, using the common ciliate *Colpoda steinii*. *Biology and Fertility of Soils* **16**, 282–286.
64. Wang, W. and Keturi, P.H. (1990). Comparative seed germination tests using ten plant species for toxicity assessment of a metal engraving effluent sample. *Water, Air, Soil Pollution* **52**, 369–376.
65. Wang, W. (1990) Toxicity assessment of pre-treated industrial wastewater using higher plants. *Research Journal of the Water Pollution Control Federation* **62**, 853–859.
66. Wundram, M., Selmar, D. and Bahadir, M. (1996) The *Chlamydomonas* test: a new phytotoxicity test based on the inhibition of algal photosynthesis enables the assessment of hazardous leachates from waste disposals in salt mines. *Chemosphere* **32**, 1623–1631.
67. Debus, R. and Niemann, R. (1994) Nematode test to estimate the hazard potential of solved contamination. *Chemosphere* **29**, 611–621.
68. Samoiloff, M. (1990) The nematode toxicity assay using *Panagrellus redivivus*. *Toxicity Assessment* **5**, 309–318.
69. van Kessel, W.H.M., Brocades Zaalberg, R.W. and Seinen, W. (1989) Testing environmental pollutants on soil organisms: a simple assay to investigate the toxicity of environmental pollutants on soil organisms, using $CdCl_2$ and nematodes. *Ecotoxicology and Environmental Safety* **18**, 181–190.
70. Römbke, J. and Knacker, Th. (1989). Aquatic toxicity tests for enchytraeids. *Hydrobiologia* **180**, 235–242.
71. Joern, A. and Hoagland, K.D. (1996) In defence of whole-community bioassays for risk assessment. *Environmental Toxicology and Chemistry* **15**, 407–409.

72. Davis, R.D., Beckett, P.H.T.and Wollan, E. (1978). Critical levels of twenty potentially toxic elements in young spring barley. *Plant and Soil* **49**, 395–408.
73. van Straalen, N.M. (1996) Critical body concentrations: their use in bioindication. In: *Bioindicator Systems for Soil Pollution* (N.M. van Straalen and D.A. Krivolutsky, eds). Kluwer, Dordrecht, pp. 5–16.
74. van Wensem, J., Vegter, J.J. and van Straalen, N.M. (1994) Soil quality criteria derived from critical body concentrations of metals in soil invertebrates. *Applied Soil Ecology* **1**, 185–191.
75. Bayley, M., Baatrup, E. and Bjerregaard, P. (1997). Woodlouse locomotor behaviour in the assessment of clean and contaminated field sites. *Environmental Toxicology and Chemistry* **16** in press.
76. Sorensen, F.F., Weeks, J.M. and Baatrup, E. (1997) Altered locomotor behaviour in woodlouse (*Oniscus asellus* L.) collected at a polluted site. *Environmental Toxicology and Chemistry* **16**, 685–690.
77. Peakall, D.B. and Shugart, L.R. (1993) *Biomarkers. Research and Application in Assessment of Environmental Health.* Springer-Verlag, Berlin.
78. Hugget, R.J., Kirmle, R.A., Mehrle, O.M. and Bergman, H.L. (1992) *Biomarkers. Biochemical, Physiological, and Histological Markers of Anthropogenenic Stress*. CRC Lewis, Boca Raton, FL.
79. Peakall, D.B. (1994). Biomarkers. The way forward in environmental assessment. *Toxicology and Ecotoxicology News* **1**, 55–60.
80. Scott-Fordsmand, J.J. and Weeks, J.M. (1998) *A Review of Selected Biomarkers in Earthworms. Recent Advances in Earthworm Ecotoxicology*. SETAC Press.
81. Svendsen, C. and Weeks, J.M. (1997). Relevance and applicability of a simple earthworm biomarker of copper exposure. I. Links to ecological effects in a laboratory study with *Eisenia andrei*. *Ecotoxicology and Environmetal Safety* **36**, 72–79.
82. Svendsen, C. and Weeks, J.M. (1997) Relevance and applicability of a simple earthworm biomarker of copper exposure. II. Validation and applicability under field conditions in a mesocosm experiment with *Lumbricus rubellus*. *Ecotoxicology and Environmental Safety* **36**, 72–79.
83. Verschaeve, L. and Gilles, J. (1995) Single cell gel electrophoresis assay in the earthworm for the detection of genotoxic compounds in soils. *Bulletin of Environmental Contamination and Toxicology* **54**, 112–119.

84. Hamers, T., Notenboom, J. and Eijsackers, H.J.P. (1996) Validation of laboratory toxicity data on pesticides for the field situation. RIVM Report no. 719 102 046 (plus annexes).
85. Parmelee, R.W., Wentsel, R.S., Phillips, C.T., Simini, M. and Checkai, R.T. (1993) Soil microcosm for testing the effects of chemical-pollutants on soil fauna communities and trophic structure. *Environmental Toxicology and Chemistry* **12**, 1477–1486.
86. Pontasch, K.W., Niederlehner, B.R. and Cairns, J. (1989) Comparison of single-species, microcosm and field responses to a complex effluent. *Environmental Toxicology and Chemistry* **8**, 521–532.
87. Axelsen, J.A. (1997) A physiologically driven mathematical simulation as tool for extension of results from laboratory tests to ecosystem effects. In: *Ecological Risk Assessment of Contaminants in Soil* (N.M. van Straalen and H. Løkke, eds) Ecotoxicology Series vol. 5. Chapman & Hall, London, pp. 233–252.
88. Clements, W.H. and Kiffney, P.M. (1994) Assessing contaminant effects at higher levels of biological organisation. *Environmental Toxicology and Chemistry* **13**, 357–359.
89. Kimball, K.D. and Levin, S.A. (1985) Limitations of laboratory bioassays: the need for ecosystem-level testing. *BioScience* **35**, 165–171.
90. van Straalen, N.M. and Løkke, H. (1997) Ecological approaches in soil ecotoxicology. In: *Ecological Risk Assessment of Contaminants in Soil* (N.M. van Straalen and H. Løkke, eds) Ecotoxicology Series vol. 5. Chapman & Hall, London, pp. 3–21.
91. Blanck, H. and Wängberg, S.Å. (1988) Induced community tolerance in marine periphyton established under arsenate stress. *Canadian Journal of Fisheries and Aquatic Science* **45**, 1816–1819.
92. Díaz-Raviña, M., Bååth, E., and Frostegård, Å. (1994) Multiple heavy metal tolerance in soil bacterial communities and its measurement by a thymidine incorporation technique. *Applied and Environmental Microbiology* **60**, 2238–2247.
93. Rutgers, M., van´t Verlaat, I., Wind, B., Posthuma, L. and Breure, T. (1997). Rapid method to assess pollution-induced community tolerance in contaminated soil. Poster at the SETAC Conference, April 1997, Amsterdam.
94. Axelsen, J.A., Strandberg, B., Bruus Pedersen, M. and Jensen, J. (in preparation) Biomonitoring of polluted sites by use of plant and soil fauna analysis.
95. Axelsen, J.A., Krogh, P.H., Holmstrup, M. and Jensen, J. (in preparation) The impact of a DDT gradient on the structure of microarthropod communities in forest soils.

96. Pedersen, M., Axelsen, J.A., Strandberg, B. and Attril, M. (in preparation) The impact of a copper gradient on the structure of micro-arthropod communities.
97. Strandberg, B., Axelsen, J.A., Bruus Pedersen, M. and Attril, M. (in preparation) The impact of a copper gradient on plant community structure.

Chapter 5

Site and source characterisation

5.1 Introduction

Risk assessment of contaminated sites depends fundamentally on the data available about the characteristics of the site, including contaminant sources, pathways and the potential receptors that may be at risk. The ultimate aim of a site investigation is to provide appropriate and reliable data against which to assess risks to chosen receptors, which may include:

- humans
- the water environment
- flora and fauna/ecosystems
- buildings and structures.

Therefore site investigation is required to gather all the information about contaminant sources and expected behaviour needed for a complete assessment. Contaminants may be present within different media such as soils, groundwater, surface waters, sediments, flora, fauna, and indoor or outdoor air; they may be found in the solid, liquid or gaseous phases. Site investigation therefore needs to:

- determine the nature and extent of any contamination on the site, and entering or leaving the site;
- identify any other hazards needing immediate action to protect public health or environmental quality;
- identify and characterise pathways and targets (elaboration of a site conceptual model);
- determine the need for long-term monitoring;
- provide sufficient information on sources, pathways, receptors and other site-specific characteristics to permit risk assessments;
- provide sufficient information to enable selection of appropriate remedial action.

In principle, a very large number of parameters might be evaluated during a site investigation, all adding to the costs of the investigation. A careful choice, restricting the parameters to those necessary and sufficient for the assessment, is therefore of the utmost importance. In addition, questions about the spatial scale of sampling and the accuracy and precision of measurements need to be specified. In general the data issues that may affect the confidence that can be placed in the resulting estimates of risk are:

- sufficiency or completeness, in scope and quantity;
- relevance to risk assessment;
- accuracy and reliability;
- handling of ambiguities and uncertainties.

It is important to recognise that site investigation for contaminated land risk assessment purposes is only one type of investigation that may be required for any particular site. Depending on the overall reasons for investigating a particular site (such as environmental concerns, reclamation or redevelopment, regulatory intervention, or commercial land transactions) data may be required for:

- hydrological or hydrogeological studies;
- assessing risks to human and ecological receptors;
- geotechnical assessment;
- planning of construction or engineering works;
- ecological assessment.

In some cases investigations are carried out solely for contaminated land risk assessment purposes; in others, one or more of the investigations may be combined. The approach adopted can affect the detailed strategies and methodology adopted.

The extent to which site investigations for risk assessment, and for identifying, evaluating and defining remedial strategies, are combined varies from country to country. Almost all countries recommend that, if a site investigation is not carried out solely for the purpose of contamination risk assessment, then the risk assessment objectives should not be compromised by the need to combine different types of investigation.

These objectives will best be achieved if the methodology for site and source characterisation is seen to be part of an explicit data quality planning (DQP) process. DQP is a tool that will increase the likelihood of collecting appropriate and useful data cost effectively, and hence will streamline risk assessment. It also provides a convenient way of documenting activities and decisions, and fosters effective communication among the various stakeholders. These are central tenets of quality management practices.

DQP involves six key steps:

1. **Define the study question** so that the focus of the investigation is not ambiguous.
2. **Define the decision statement** that the investigation will attempt to resolve. The decision statement links the study question to possible outcomes.
3. **Identify the inputs** required to resolve the decision statement. This includes the area of land to be sampled, and whether separate designations of parts of a larger area of land may simplify the investigation; the types of sample; the measurements that may be required; the sources of other required data; the modelling requirements (if any); and the practical constraints on data collection.
4. **Develop a decision rule or rules**, i.e. the logical basis for choosing between alternative actions. This is likely to be in the form of one or more *if … then* statements related to statistical parameters (e.g. mean, maximum, percentile value) derived from the analytical dataset.
5. **Specify limits on decision errors**, which will reflect the tolerance for errors based on consideration of the consequences of making an incorrect decision (see Chapter 2, Section 2.3).
6. **Identify the most cost-effective design** for collecting the data that are expected to satisfy the other DQP objectives. This will include deciding whether intrusive investigations should be conducted in more than one phase. It might also include use of surrogate measures (e.g. total petroleum hydrocarbons) or composite samples.

5.2 Current practice

During the CARACAS project the various approaches to contaminated site investigation in participating countries have been reviewed. In order to compare these approaches and to make recommendations for further research and development, four main themes have been identified which can serve as focal points for discussion:

1. Data needs for risk assessment, i.e. the relationship between site investigation and risk assessment
2. Strategies for data collection
3. Methodologies for data collection
4. Data quality and uncertainties.

Strategy refers to the types of data required and the judgements made on where to acquire the data; *methodology* refers to the

techniques used to collect data. Two general types of national procedure have been identified. Some countries have developed or are about to develop general procedures including investigation and data collection strategies based on best available practices. These practices are in constant evolution, and documents need to be updated periodically. Other countries only define the general objectives of site investigation in a risk assessment context. Detailed procedures are left to the discretion of the practitioner.

Data needs for risk assessment

Historically there has been a tendency to carry out site investigation without explicitly relating the investigation to risk-based objectives and needs. This is particularly the case in countries where much site investigation was focused on the data needs for construction or development work. Today, however, all the participating countries view contaminated site investigation as a tool intimately linked to the risk assessment process. A phased approach is generally seen as providing the required data in the most cost-effective and efficient manner. The phased approach typically comprises:

1. Preliminary investigation (desk study, site reconnaissance and sometimes limited exploratory investigation). The goal of this preliminary stage is to assess whether potentially contaminating activities have taken place on the site, whether soil and/or water pollution is suspected, and in some cases to confirm the existence of pollution. In short, this phase focuses on hazard identification.
2. Detailed investigation. The aims at the main site investigation stage are (i) to define the extent and degree of contamination, (ii) to assess the risks associated with identified hazards and receptors and (iii) to determine the need for remediation in order to reduce or eliminate the risks to polluted or actual receptors.
3. Supplementary or feasibility investigations to better define the need for and type of remedial action or monitoring. The aim may be to assess the feasibility of various remediation techniques; this may include more detailed physical and chemical characterisation of soils and laboratory studies on soil or groundwater treatability. Supplementary investigations may also be designed to improve understanding of the nature, extent and behaviour of contaminants.

Current practices for acquiring the data needed for risk assessment appear to be relatively consistent across the participating countries, although some differences in phasing or terminology are obvious.

Most countries recommend that risk-related data needs are explicitly identified before starting investigative work, and are reviewed and refined at the start of each new phase of investigation as part of the objectives for the work. Similarly, in the majority of cases the normal practice for site investigation involves developing a conceptual model about the site which identifies source–pathway–receptor relationships. This model can then be developed and refined as additional data are obtained.

Data for risk assessment are required on the following topics:

- Ground conditions, including the data needed to construct a preliminary hydrological model;
- Physical and chemical characteristics of contaminants, including their spatial distribution and expected transport and fate;
- Ecological characteristics of the site;
- Information relevant to public and worker safety.

Data needs for subsequent modelling must also be considered in relation to:

- Human toxicology (for example, acute and chronic toxicity, mineral and chemical speciation);
- Ecological risk (for example, toxicity and potential for bioaccumulation and biomagnification);
- Transport and fate (for example, pH, redox potential, hydraulic conductivity etc.).

Strategies for data collection

To ensure that resources are used to best effect and that the necessary data is obtained, an investigation strategy should be developed. Of particular importance are sampling strategies and analytical strategies (for both on-site and laboratory testing).

Most countries have developed standard procedures or protocols for data collection but in most cases the procedures will not be fully implemented until 1998, nor will they necessarily be mandatory. In Flanders (Belgium) these protocols have been mandatory for recognised experts since 1997, but well-motivated changes are always permitted. In the Netherlands, they are also mandatory, although alternative approaches are allowed if one can show that the quality of an investigation is likely to be equal or better than one conducted using the protocols.

In Germany the analysis and evaluation of risk from contaminated sites is normally governed by regulations and technical guidance issued by the various Länder (federal states). The Länder have different

strategies for performing site investigations and use different analytical methods. Harmonised procedures are being developed within the framework of the Länder working party on quality assurance.

The following aspects of data collection strategy have generally been adopted by all participating countries:

- The information needed to estimate and evaluate risks;
- Any information already available about sources, pathways and receptors;
- The degree of confidence required of the risk estimates;
- The techniques and methods which are suitable and likely to be most effective in collecting information of appropriate quality;
- The screening approaches or models to be used in risk estimation and evaluation;
- The type(s) of receptor potentially at risk;
- The media in which contaminants may be found.

There are some exceptions, however. For example, in Austria screening approaches and the types of receptor potentially at risk are not taken into account in the data collection strategy.

Most participating countries have made similar decisions about which types of receptor to consider, namely humans, surface water and groundwater, flora and fauna, and buildings. Receptors are identified on a site-specific basis. However, it is not routine to collect specific data about receptors and their behaviour for a particular site. In most countries this would only be necessary if more detailed studies were judged necessary.

Sampling strategies are recommended in all countries; in some countries (e.g. Austria) sampling strategies are only provided for groundwater while in others they have been developed with particular reference to human health. In the UK, recommended soil sampling strategies for human health risk assessment have an explicit statistical basis. In the Netherlands sampling recommendations have an underlying statistical basis but also take cost into account. In contrast to sampling strategies, national analytical strategies have not been developed in most countries. The analytical methods used, and the level of quality control and quality assurance, are left to professional judgement.

It is important to emphasise that there are no hard and fast rules for the number of samples to be collected and analysed. However, the data quality planning process described at the end of Section 5.1 provides a structured approach for designing cost-effective sampling strategies, which ties sampling and analytical design back to the study question(s)

and decision statement(s). Sampling strategies also need to be flexible so that assessors can respond to new information that becomes available, or to unanticipated constraints on data collection.

Methodologies for data collection

These include the following:

- Desk-based studies
- General guidance on site investigation methodology
- Sampling techniques (intrusive and non-intrusive)
- Non-intrusive investigation techniques, e.g. geophysics
- Analysis (both on-site and laboratory)
- Data presentation.

A wide range of procedural and technical guidance is currently used by contaminated land practitioners. The current situation among participating countries suggests that, whereas technical guidance on site investigation methodology is widely available, it is not always focused on *contaminated* sites, and broader geotechnical issues are often covered in more detail.

In addition to government-sponsored work, guidance has been or is being developed by specific industries (e.g. the petroleum industry) and also, in the UK, by various public sector development agencies, private sector organisations and professional bodies. This guidance tends to be general in nature, addressing underlying issues and overall approaches.

However, there are no standard protocols for sampling in any of the participating countries, with the choice of sampling methods being left to practitioners, who are expected to choose appropriate sampling techniques for the circumstances. ISO documents and relevant national standards (e.g. AFNOR, BSI, DIN, SS) are widely used as sources of detailed information on sampling techniques and other methodologies.

A common approach among countries is also evident in choosing the contaminants to be analysed. Selection of contaminants is generally based on consideration of previous site use and perhaps by reference to a priority list of substances (e.g. Belgium/Flanders, Spain, Sweden, UK). The basis for choosing a priority list may differ in detail from country to country, but mostly it is based on a consideration of key industrial contaminants, combined with the potential to harm human health and the environment (see Chapter 3). For example, the Flanders priority list comprises:

- for soil samples: pH; organic carbon content; clay content; toxic metals (Pb, Zn, Cd, Cu, Ni, As, Hg, Cr); mineral oil; polyaromatic hydrocarbons; EOXs
- for groundwater: pH; conductivity; toxic metals (Pb, Zn, Cd, Cu, Ni, As, Hg, Cr); benzene, toluene, xylene; mineral oil; volatile organic compounds.

In Sweden, the standard list focuses on metals and petroleum hydrocarbons. In the Netherlands, some contaminants are always analysed (for example, metals and PAHs)

Laboratory test methods typically comprise DIN, ISO and USEPA methods and, in some countries, are specifically related to guideline values. Also, some countries consider it good practice to specify methods whereas others prefer laboratories to select methods that are *fit for purpose* (e.g. France, UK). Although laboratory test methods for most metals are considered adequate, further method development on organics (e.g. phenols, polyaromatic hydrocarbons and total petroleum hydrocarbons) and some inorganics (e.g. cyanide and sulphide) is felt necessary by participating countries.

On-site analysis may be preferred for some determinands, and for major investigations mobile laboratories may be used. There is much interest in using rapid on-site methods (e.g. immuno-assay techniques) to help guide detailed sampling design; but this approach is not routinely used in any of the participating countries.

Current practice on reporting data from site investigations is for national or sector-specific guidance to set out minimum reporting requirements for different phases of an investigation. It is generally felt that data interpretation can be performed by staff from a range of disciplines providing they have appropriate scientific qualifications and relevant experience.

Data quality and uncertainties

Data from site investigation is fundamental to risk assessment of contaminated sites. Hence it is vital to ensure that the data is representative of the site and is of suitable quality. General guidance on quality assurance in sampling, including sample handling, transport, storage and chain of custody documentation, is typically provided by participating countries.

Quality assurance for laboratory analysis is provided by all CARACAS countries through method validation, internal quality control procedures (e.g. using certified reference materials where available) and external quality control procedures (e.g. via independent

accreditation or participation in inter-laboratory comparisons). Current practice indicates that quality assurance approaches can benefit site investigation for contaminated land risk assessment, but that historically insufficient attention has been paid to them.

Estimation of uncertainties is of crucial importance to risk evaluation as it provides a measure of confidence in the site investigation data. Uncertainties typically concern:

- The extent to which sampling techniques are adequate to ensure that a sample is representative of the site conditions at the point where it was taken;
- The extent to which contaminant data from single samples are spatially representative of the site conditions;
- The extent to which the analytical data reflects actual characteristics of the contaminants present, e.g. concentration, speciation, mobility, availability etc.;
- The extent to which the geological, hydrogeological and geotechnical conditions at a site are understood;
- The ways in which contaminants behave in the environmental setting of the site and surrounding areas (fate and transport issues);
- The extent to which behaviour of the receptors potentially at risk under the particular circumstances may affect the risk estimates;
- How the receptors may be affected by contaminants, and what role different receptor characteristics play in this.

Uncertainties therefore relate to all aspects of site investigation, including the design of the investigation and the methods or techniques used to collect the data.

Uncertainties have not always been considered explicitly. Recent UK guidance, for example, has drawn attention to this aspect and the forthcoming model procedure for risk assessment explicitly addresses data uncertainties, their effect on risk estimates and the importance of evaluating such uncertainty in a formal way. However, in none of the participating countries is there a requirement at present to specify confidence limits on site investigation data. Advice on statistically valid sampling for soils has been developed in the UK, where the confidence level normally used to illustrate the statistical basis of a sampling strategy is 95% confidence of detecting a contaminant hotspot of a given size and shape.

With regard to laboratory analytical data, it is not common for error ranges to be requested by those specifying laboratory testing, nor to be reported or taken into account in risk assessments. However, good practice in quality assurance suggests that they should be reported.

5.3 Concluding remarks

Investigation of a site is an essential task to enable appropriate data to be obtained for risk assessment. A site investigation should be viewed as a process of continuous phased exploration and interpretation, with the scope of the investigation being adjusted in the light of the data obtained. The key scientific aims for site investigation and analysis are:

1. The data obtained should be reliable, reproducible and representative of the actual site conditions. This aim refers to the quality of the data obtained from investigations, both at the site and in the laboratory.
2. The data obtained should be relevant and sufficient for risk assessment purposes. Thus data are required concerning:
   (i) the location, extent and types of contamination expected to be present;
   (ii) the geological *and geochemical* conditions of the site and surrounding land;
   (iii) the hydrogeological and hydrological regime for the area;
   (iv) the known anticipated presence and behaviour of receptors.
   On some occasions, ecological and ecotoxicological information may also be required.
3. The methods used for obtaining the data should be appropriate to the conditions of the environmental medium and the contaminants. Interpretation of data is influenced by the uncertainties and limitations inherent in the methods used. Therefore these need to be understood and quantified whenever possible.
4. The uncertainties and assumptions inherent in, or associated with, the data should be capable of being evaluated in the context of risk assessment.

The data quality planning (DQP) process described in Section 5.1 offers a structured approach for data collection, which should help to ensure that site investigations are properly focused and transparent in their objectives. Ideally, all the relevant stakeholders should *buy in* to the site investigation strategy at the DQP stage so that trade-offs between cost and confidence are clear at the outset.

In compiling the information on which this chapter is based, a considerable database of technical and procedural guidance documents produced by the participating countries has been assembled. This will be published in Volume 2, *Risk Assessment for Contaminated Sites in Europe: Policy Frameworks.*

Chapter 6

Pathways: transport and fate of contaminants

6.1 Introduction

The role of fate and transport in contaminated site risk assessment is illustrated in Figure 6.1. In the context of the source–pathway–receptor model for risk assessment, the study of *fate and transport* is mainly concerned with describing and understanding the various pathways or routes through which a receptor might be placed at risk from contamination. The receptor might be people, ecosystems, groundwater resources, buildings etc.

Where receptors are not directly exposed to a contaminant, risk assessment needs to consider the various ways by which indirect exposure might occur and their significance. Routes through which a contaminant might be transported include soil, groundwater, surface water, uptake or adsorption by plants, dust, aerosols etc. However, a contaminant may also undergo transformations through biological, chemical or physical means that might affect its toxicity, availability and mobility. In practice, our knowledge of how contaminants in a contaminated site are affected by the various fate, attenuation and transport processes is low. This means that many contaminated sites are like *black boxes* in the sense that we have little knowledge of the complex interactions that might be taking place within a site and how they are affecting the risk of receptor exposure. This is particularly true where a site is contaminated with mixtures of contaminants, as is often the case.

During transport of contaminants in soil, the contaminants are affected by a number of physical or reactive geochemical and biological processes, which may attenuate, concentrate, immobilise, liberate, degrade or otherwise transform the contaminants. The risk depends on both the concentration of the contaminant and the route of exposure

(skin contact, inhalation, ingestion, etc.). For this reason, analysis of the changes that the contaminant undergoes as a result of these transformation and phase transfer processes prior to exposure is an important part of exposure analysis and assessment. This analysis is potentially very complicated, since the number and types of processes affecting the contaminants during the transport is governed by both inherent contaminant characteristics and environmental conditions. In addition, the presence of some contaminants might change the environmental conditions. One example is that the introduction of biodegradable contaminants will stimulate microbial activity, leading to depletion of electron acceptors and change in redox conditions; the redox conditions in turn have a major impact on the biodegradation of other compounds. Understanding these complex dynamic processes requires a detailed knowledge about the environmental chemistry of contaminants (biodegradability, hydrophobicity, etc.) and the environment at the site (geology, geochemistry, etc.).

The potential processes involved in fate and transport depend on the types of contaminant and include:

- Biodegradation
- Sorption/desorption
- Binding/incorporation

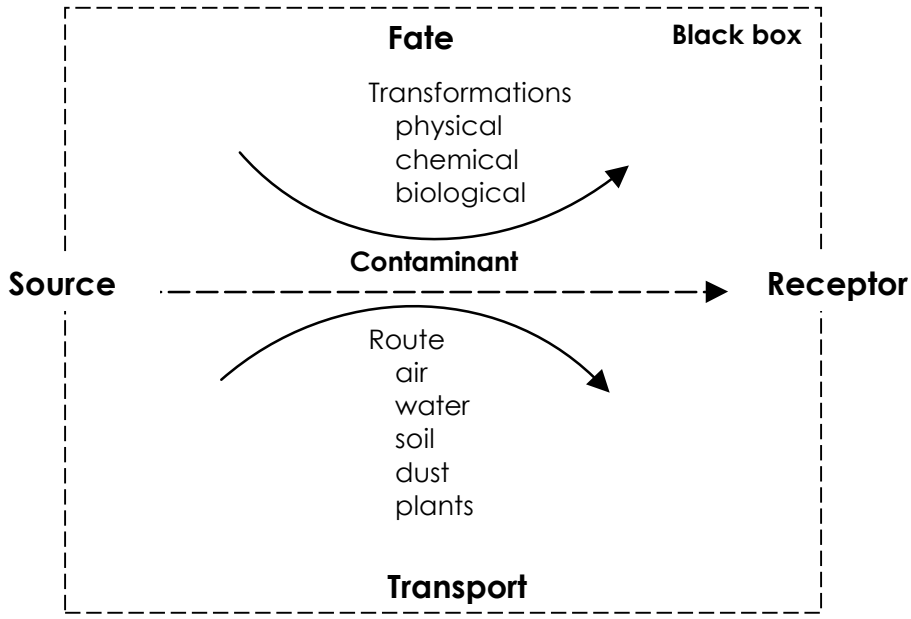

Figure 6.1 The role of fate and transport in risk assessment of contaminated sites

- Dispersion
- Solubilisation
- Diffusion, including intraparticle diffusion
- Complexation
- Precipitation/dissolution
- Evaporation
- Chemical oxidation
- Photo-oxidation
- Plant uptake

Fate and transport analysis does not normally include all the above processes, but it must at least integrate the key processes. Whenever possible, a quantitative approach should be applied in risk assessment. This requires a reliable mathematical description of the processes as a function of changes in geochemical, geological and biological conditions. Often the soil environment is characterised by high complexity and inhomogeneity, which makes it even more difficult to estimate contaminant fate and transport.

A number of models have been developed which attempt to integrate some of the processes in question, mostly biodegradation, sorption and dispersion for organic contaminants in groundwater (MIKE SHE, MODFLOW, etc.) and/or speciation-based equilibration reactions for toxic metals (GEOCHEM, PHREEQUE, MINTEQ). Most of the models used for organic compounds do not take into account all the different partitioning processes in a soil system. Many of them often simplify the mathematical description of processes (e.g. biodegradation by first order kinetics) in order to handle them mathematically; and none has been developed to a point where all the different phases of a soil system (soil particles, organic matter, free phase product, pore water, pore vapours) are adequately addressed.

This chapter includes a general overview of some of the key issues associated with transport and fate of contaminants. It reviews in Section 6.2 some of the key processes involved in the transport and fate of contaminants in groundwater. No references are provided for this part of the chapter because the literature in this area is enormous; the emphasis instead is in setting out some of the principles. Readers wishing to access the literature are advised to consult recent issues of specialist journals (e.g. *Journal of Contaminant Hydrogeology*, *Ground Water*, *Water Resources Research*) and textbooks.[1,2,3]

The chapter continues with a very brief section (6.3) on transfer of contaminants from soil to surface waters, and a review (Sections 6.4–6.6) of contaminant exposure via fugitive dust, soil-vapour intrusion

into buildings, uptake by plants, and direct soil ingestion. Further details of several aspects are given in Chapter 7, 'Models'.

The overriding scientific aim of fate and transport research and development is to improve our level of understanding of how contaminants are affected by the many physical, chemical and biological processes that occur within a contaminated site (i.e. lighting up the *black box*). A research and development framework is proposed in Figure 6.2, which illustrates the connection between medium to long-term research at both fundamental and applied levels and the development, in the short to medium term, of site-specific tools to help risk assessment practitioners.

An immediate priority is the development of guidance on the role of fate and transport in risk assessment of contaminated sites. This guidance should concentrate on describing and explaining data and information needs so that the role of fate and transport processes at a site, and their impact and effect on receptor exposure, can be determined. The products of medium to longer term research would then feed into such guidance so that risk assessment practice can be continually improved as our understanding of what happens in the black box improves.

It should also be emphasised that there is considerable overlap

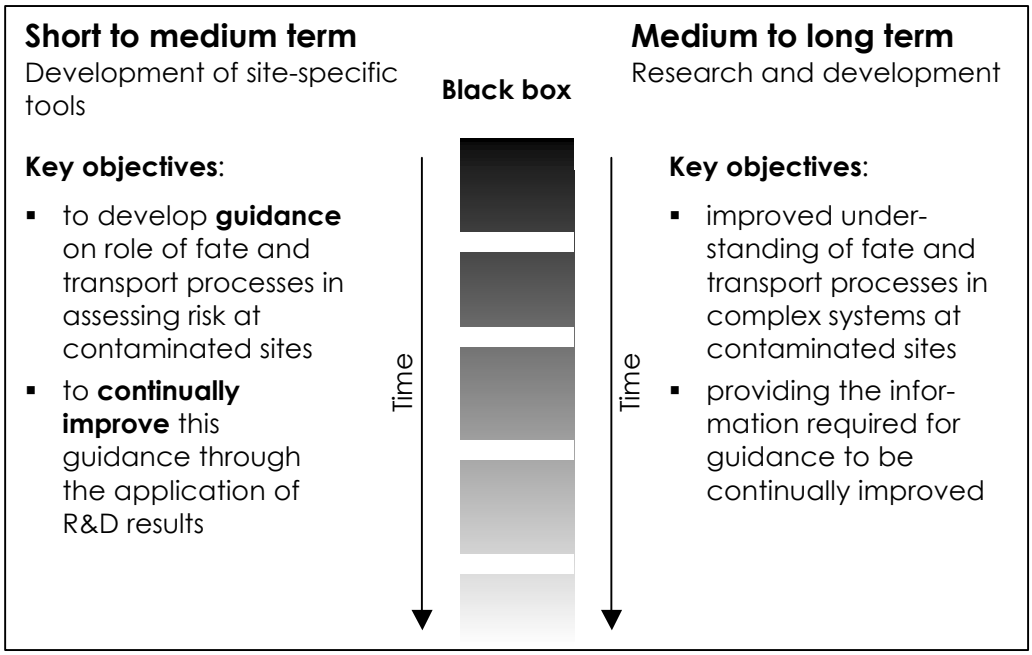

Figure 6.2 Proposed research and development framework

between the type of fate and transport information needed to assess risk and the type of information needed to design *in situ* remediation.

6.2 Transport and fate in groundwater

This section describes current *fate and transport* principles used in assessing groundwater pollution risks from contaminated sites. To provide some insight into the ways of quantifying fate and transport processes in a risk assessment, the simple mathematical descriptions applied in current research and risk assessment models are given in the sections below. Only simple expressions are included. However, some studies have applied much more advanced expressions, but they have not yet found their way into most risk assessment models. Likewise, some of the processes listed earlier, e.g. diffusion, are not included below despite their scientific importance. The reason is that they are not usually significant in risk assessments.

From the source area, which at many contaminated sites is the soil at or just beneath the surface, contaminants are transported vertically down to groundwater either as free product or as solutes in the infiltration water. After having reached the aquifer the contaminants spread horizontally in the flow direction, mostly in soluble form as contaminant plumes.

Solute transport

The transport of contaminants is calculated on the basis of advection, expressed by the equation for the linear flow rate for a porous medium:

$$J_{adc} = Q/A \times C_w \tag{6.1}$$

or by the so-called average pore water velocity; that is, the flow velocity in the pores:

$$v_p = v/\varepsilon$$

The velocity is determined by the hydraulic gradient dh/dx and the hydraulic conductivity K of the soil:

$$V = -K(dh/dx) \tag{6.2}$$

This is Darcy's law, in which the hydraulic gradient relates to the change in head or hydraulic potential dh over the distance dx in the flow direction.

Contaminants that are transported with the water in soluble form are subject to both phase-transfer processes, as described above, and dispersion. In many cases Fick's law is applied to describe the macroscale dispersion. The velocity in fractures can also be described using a version of Darcy's law in which the hydraulic conductivity for porous media is replaced by a fracture hydraulic conductivity.

During transport a number of physical, chemical and biological processes act on contaminants. As contaminants move through aquifer soils and rocks, they will partition between the different phases (water, air, organic matter, free product), thus governing the contaminant concentration in groundwater. The resulting transport will then be defined as a function of partitioning between phases and water velocity. The main phase-transport and partitioning processes occurring in soil and groundwater during contaminant transport from the source area to the receptor(s) are solubilisation/extraction from free phase product, sorption/desorption, and for toxic metals complexation and precipitation. The distribution of contaminant between phases is often expressed by the concentration of the contaminant in each phase. The total concentration C_T can be expressed by:

$$\rho\, C_T = \rho\, C_s + \varepsilon_w C_w + \varepsilon_a C_a + \varepsilon_i C_i \qquad (6.3)$$

where ρ is the soil density and ε are the volumetric contents. The subscripts s, w, a and i refer to soil solids, soil water, soil air and free product, respectively.

The most frequent phase-transfer processes mentioned above are also the most frequent processes included in exposure analysis schemes. Apart from the phase-transfer processes, dispersion and degradation will also govern contaminant concentrations downstream from the source area. The different specific processes involved are described below.

Dispersion

A certain spreading or dispersion of contaminants occurs during transport to and via the groundwater. The most significant spreading is due to geological heterogeneities in the aquifer leading to macroscale aberrations from the flowpath. Normally microscale dispersion due to diffusion can be ignored. To describe dispersion, Fick's first law is normally applied. It is expressed as follows:

$$J_{disp} = \varepsilon \times D(\mathrm{d}C_w/\mathrm{d}x) \qquad (6.4)$$

where J_{disp} is the contaminant flux and D is the dispersion coefficient.

Solubilisation

If there is free product in the source area, solubilisation will be the first process to occur prior to further transport downstream. The physical contact between flowing water and the product governs solubilisation. If the product is solid and the water is flowing around the product particles, the process will be a diffusion-governed solubilisation. If the product particles are small, and water flows through the product, it will be an advection-governed solubilisation. Simple mathematical descriptions have been derived for both types of solubilisation, which can be applied for calculating water concentrations in the source area. These mathematical descriptions are based on the assumption that there is an equilibrium between water and product. However, in many cases the contact between free product and water is limited because of the physical environment, which then has to be taken into account.

When there is free product in the soil, the contaminant concentration in adjacent groundwater will be controlled by the aqueous solubility of the contaminant. Water and air close to the free source will contain contaminant concentrations up to the limit of saturation in the medium. Downstream of the source area, dispersion and degradation will then reduce contaminant concentration in the soil water and air. Practitioners often do not attempt to model solubilisation or evaporation from free product. Instead, the water and soil gas concentrations are measured directly, and these measurements then provide the basis for risk assessment.

Sorption/desorption

The partitioning of organic and many inorganic contaminants between soil and water is mainly governed the sorption/desorption process. The distribution of toxic metals is governed primarily by complexation, precipitation, and sorption. Sorption can be expressed by a simple partitioning coefficient, K_d, called the distribution coefficient which is given by the equation:

$$K_d = C_s/C_w \tag{6.5}$$

where C_s is the sorbed concentration on solid particles and C_w is the soil water concentration.

With respect to organic contaminants, K_d can be calculated using the organic carbon–water partition coefficient of the compound K_{oc} and the fraction of organic carbon in the soil f_{oc}:

$$K_d = K_{oc} \times f_{oc} \quad (6.6)$$

Distribution coefficients for toxic metals can be determined experimentally. Such studies have shown that pH is often the most important factor controlling sorption of toxic metals in soils. This suggests a relatively simple relationship between mobility of some toxic metals and the pH of the soil or aquifer.

Complexation

Formation of dissolved ion pairs in the water phase by complexation of metals with inorganic and organic complexing agents (ligands) is an important process that may affect the solubility and mobility of toxic metals. Since complexation typically involves one metal ion (M) and one ligand (L) the reaction of formation of the complex can simply be expressed (ignoring the charge of each ion) as:

$$M + L = ML \quad (6.7)$$

and the related mass interaction equation:

$$[ML]/[M] \cdot [L] = K_c \quad (6.8)$$

where K_c is the stability constant.

Complexes are common in relation to contaminated sites because the concentrations of ligands found under these conditions are relatively high. The relevant ligands are inorganic salts (chloride, sulphate and carbonate), specific complexing agents with very high K_c (e.g. EDTA) and dissolved organic molecules, including general organic matter.

Stability constants are available for inorganic and specific organic ligands with respect to most metals, and since the complex reactions are relatively fast (seconds) reliable calculations can be made. However, reactions with general organic matter (e.g. fulvic and humic substances) are more difficult to calculate and must, if possible, be supported by direct measurements of the degree of complexation.

Precipitation

Ignoring the charge on each ion, precipitation and dissolution processes can simply be expressed as:

$$MA = M + A \quad (6.9)$$

where MA is the solid phase (mineral) formed from the metal M and the anion A. Since the activity of the solid phase is defined to be unity, the mass equation equilibrium is simply:

$$[M] \cdot [A] = K_{so} \tag{6.10}$$

where K_{so} is the solubility product. This expression defines the concentrations of dissolved ions under circumstances where these are controlled by a solid phase.

Precipitation is relevant for many ions but the minerals formed are typically carbonates, phosphates, sulphides and hydroxides that also take part in acid–base reactions. Many solubility products are available in the literature but the conditions under which they were determined are not always relevant to contaminated sites. In addition, precipitation and dissolution may be very slow processes, which adds uncertainty to calculations and evaluation of the importance of these processes in actual cases.

Biological processes

Of the reactive processes potentially affecting contaminant concentrations in the transport medium, only biodegradation plays an important role. In terms of organic contaminants this process is the only process degrading and thus removing the contaminants from the environment.

Bacterial sorption/uptake

Before biodegradation can take place the contaminants and the bacteria must be brought in contact with each other. Bacteria use organic contaminants as substrate. The uptake and transformation of these contaminants takes place through a series of steps, including transport of the contaminants to the membrane surface, sorption to the cell surface, uptake and passage through the membrane, and the catalytic degradation of the compound within the cell. Bacteria then play a role as organic sorbents in the partitioning process since they have properties comparable to other types of organic matter in soil. In some environments the bacterial biomass might constitute a significant proportion of the total organic matter. In addition to the hydrophobic character of bacteria, they also possess electrophobic properties, which will affect ions, e.g. some toxic metals, that are brought in contact with the cell.

Biodegradation

Saturation kinetics provide the most precise quantitative description of biodegradation of organic contaminants. However, in nearly all exposure assessment models, simple first-order degradation kinetics are applied for the mathematical description of biodegradation. This approach is a reasonable approximation; it applies in most cases but in situations with multiple substrates, with possible substrate interactions and/or with high concentrations of the substrates, it might not adequately reflect reality. Simple first-order kinetics are based on the assumption that the concentration of the contaminant is very low. This means that the Monod constant for the contaminant $K_{cs} > S_{cs}$ (bio-available contaminant concentration), and that the electron acceptor (i.e. O_2) is in surplus, which implies that the Monod constant for the electron acceptor $K_{ac} << S_{ac}$. Under such conditions biodegradation can be expressed by the following equation:

$$R_s = kS \tag{6.11}$$

where R is the biodegradation rate and k is the first-order biodegradation rate.

To determine whether first-order kinetics can be applied in a specific case, a degradability test using a sample from the site as inoculum should be carried out. It is not enough to have data on concentrations of the contaminants and compare them with kinetic parameters from the literature, since adaptation to the contaminants at the site might have a crucial impact on biodegradability.

6.3 Transport from soil to surface waters

Transfer of contaminants from soil to surface waters is highly site-specific and depends on run-off volume, peak flow rate, soil erodability, slope length and steepness, sorption capacity of the soil, vegetation cover type, and distance to receiving body. Modelling this complex set of interrelated processes is very difficult and, in practice, surface water pollution is monitored by direct measurement. It is important, therefore, for risk assessors to anticipate the sorts of action that might release surface contaminants to surface waters (e.g. practices that take place during site redevelopment). Appropriate protective measures can then be put into place.

On some sites old service trenches, drains and other man-made pathways can form conduits allowing rapid transfer of soil pollutants to

surface water bodies. Contaminants can, of course, also enter surface waters via groundwater baseflow.

6.4 Transport and fate in air

Transport via dust

Fugitive dust is dust released from soil by the action of wind, with or without the assistance of mechanical disturbance. Under dry conditions, very high releases of dust may occur during heavy construction operations or when vehicles are moving over bare soil.

There is no real consensus within the scientific community on how best to model fugitive dust emissions. It is clear that the amount of contaminant inhaled with dust will depend on the size and shape of the source area; the wind/speed and wind direction; the soil type and its surface roughness and moisture content; the degree of surface compaction; the nature, extent and quality of vegetative cover; and the locations of receptors relative to the source and the wind direction.

In addition, it needs to be recognised that exposure to soil contaminants via fugitive dust will also be influenced by some other factors:

- some inhaled dust will be derived from non-site sources which can usually (but not always) be assumed to be contaminant free (i.e. the average contaminant concentration will be diluted by non-site dust);
- fine-grained particles are easier to mobilise as fugitive dust; they can also be expected to be somewhat enriched in contaminants relative to larger particles;
- only dust particles less than 10 μm in diameter (PM_{10}) are usually assumed to be respirable; larger particles are trapped in the upper respiratory tract, from where they are expectorated or swallowed;
- the bioavailability of soil contaminants entering the body attached to soil particles is poorly understood, and will depend on whether transfer to the systemic circulation is via the lung or the gut. Some contaminants are well recognised as inducing different effects depending on the route of entry into the body.

Outdoor soil can also be carried into buildings via footwear, clothing, toys and pets and by airborne dust resuspended from local soil. Its contaminant concentration will be diluted by other sources of household dust, such as fabrics, powders, skin flakes, insects and smoke. Exposure to indoor dust also depends on the frequency of activities that liberate dust from the various surfaces on which it settles, and the extent to which dust is allowed to accumulate. These complexities simply empha-

sise that generic *modelling* of exposure to fugitive dust is all but impossible. There are, for example, wide variations in observed relationships between indoor dust and outdoor soil contaminant concentrations. Keenan *et al.*[4] assumed that the proportion of locally derived soil particles in indoor dust was 75%, based mainly on a study of land contamination around a single smelter. However, Murphy *et al.*[5] point out that in mining communities the relationship between indoor and outdoor dust appears to differ from that in smelter communities.

For example, in a USEPA study[6] at Park City, Utah, where there is widespread arsenic contamination in mine tailings, linear regression of indoor (C_i) against outdoor (C_o) arsenic concentrations in dust yielded the following:

$$C_i = 0.13\ C_o + 4.5 \text{ mg kg}^{-1} \quad (6.12)$$

This can be interpreted as an outdoor-to-indoor transfer coefficient of 13% plus a constant corresponding to indoor sources of arsenic (e.g. coal combustion, tobacco smoke). A regression analysis for former lead mining communities in England[7] yielded:

$$C_i = 0.15\ C_o + 500 \text{ mg kg}^{-1} \quad (6.13)$$

Thus the transfer coefficient is very similar but there is a much larger constant, presumably reflecting dust derived from lead paint.

In contrast to these results Murphy *et al.*[5] drew attention to several studies showing that, in smelter communities, contaminant levels in indoor and outdoor dust are roughly the same ($C_i \approx C_o$). It is possible that the surface properties and moisture content of smelter particles allow them to adhere better to shoes, clothing and pets and thus to be more readily tracked indoors.

Transport via vapour

The transport of volatile soil contaminants into the atmosphere depends on three distinct steps:

- partitioning of the chemical from soil particles to the soil air, or from groundwater or free product to the soil air;
- flux of the chemical from soil air to the atmosphere;
- atmospheric dilution of vapour in the breathing zone.

Equilibrium partitioning of organic contaminants between the solid, liquid and vapour phases of soil is controlled by three main factors: the organic carbon distribution coefficient K_{oc}, the fraction of organic carbon in the soil f_{oc}, and Henry's law constant, which in turn is determined by

the vapour pressure and aqueous solubility of the substance. Thus, referring back to Equation (6.3) if we assume no free product ($\varepsilon_i = 0$), linear adsorption ($C_s = K_{oc} f_{oc} C_w$) and liquid–vapour partitioning described by Henry's law ($C_a = H' C_w$), then Equation (6.3) can be re-arranged to give:

$$C_T = C_a \left[\frac{K_{oc} \cdot f_{oc}}{H'} + \frac{\varepsilon_w}{\rho H'} + \frac{\varepsilon_a}{\rho} \right] \tag{6.14}$$

For ionisable organic compounds the ionised and neutral species will have different partition coefficients. The overall K_{oc} value will depend on the proportions of neutral and ionised species, which are controlled by pH and the acid dissociation constant.

The flux of a volatile chemical from soil to outside atmosphere is usually assumed to follow Fick's first law of diffusion under steady-state conditions. This neglects some important processes that could affect long-term exposure (e.g. source depletion, and chemical/photochemical/-biological transformations during transport). However, for most chemicals the flux is sufficiently small, and the mixing with outside air sufficiently great, that concentrations at the breathing zone are of little significance in overall human exposure. Soil-air concentrations might, of course, be detrimental to crops or soil organisms even when the risks to human health are negligible.

Soil vapours migrating into the living and working spaces of houses and commercial properties can contribute significantly to total human exposure, and may sometimes indeed be the dominant exposure pathway. Vapour intrusion mechanisms include diffusion and pressure-driven flow induced both by temperature difference (the so-called *stack effect*) and by wind. Indoor air concentrations may be strongly influenced by soil permeability, pressure difference, floor leakage rate and the air exchange rates in living spaces and basements/crawl spaces.

Model development is relatively well advanced (see Chapter 7) but there is a great paucity of measurements for model calibration and validation. In particular, floor leakage rates and crawl spaces are both very poorly constrained, and there is a serious shortage of long-term monitoring data on indoor air concentrations of volatile organic chemicals in houses built on or adjacent to sites contaminated with VOCs.

Long-term exposure assessment is also critically dependent on the rate of source removal by volatilisation and biodegradation. The latter is highly site-specific and depends on soil type and the types and growth rates of microbial populations, which in turn are influenced by moisture content, temperature, Eh, pH, nutrient availability and toxicity. Model-

ling source decay using first-order or second-order degradation kinetics is relatively straightforward. However, as discussed earlier in this chapter, there appear to be no rapid and low-cost methods for estimating site-specific rate constants.

6.5 Transport via plant uptake

Metals

Metals in soil can occur in at least five different soil phases:

- in the crystal lattices of primary minerals;
- in the precipitation phase (carbonates, sulphides, hydroxides etc.);
- in the exchangeable phase, either attached to exchange sites on clays, hydroxides and organic matter, or complexed/chelated to organic matter;
- in the biophase;
- in the soil-solution phase.

Typically only a small percentage of the total metal content will be in the soil-solution phase, and speciation within that phase will depend on soil conditions such as pH and redox potential.

Root uptake of a metal depends on the fraction of total metal in soil that is accessible to plant roots, and the ability of the plant to transfer the metal across the soil–root interface. Availability of a metal is determined by its chemical form and its location in the soil. The most available metals are those present in soil-solutions in the ionic state or as soluble organic-matter complexes. The least available are bound firmly into primary mineral lattices.

At present there are no methods that allow for the amount of available metal in a soil to be measured directly. Analysis of soil and plant tissues can establish uptake after it has taken place but this is of limited value for predicting future uptake. Interpretation is further complicated by large differences in the ability of different plant species to uptake metals from particular soils, and by variations in metal content as a function of growth stage and plant part.

There is seldom a close or consistent relationship between *total* metal content in a soil and plant uptake. The best approach would seem to be to analyse the naturally occurring soil-solution. However, it is often very difficult to take a well-defined soil-solution sample and this problem, combined with the very low levels typically present in soil solutions, have ruled out this approach for routine analysis.

The usual practice among soil scientists is to attempt to establish empirical relationships between plant metal content and the amount of metal that can be extracted from the soil by specific reagents in the laboratory. A very wide range of reagents has been used, from water alone to aggressive solutions such as 1M HCl or acidified ammonium oxalate.

Some metals become more available to plants as pH decreases (e.g. Co, Ni, Cd), others become less available (e.g. As, Mo, Se), whereas some are only slightly affected (e.g. Cu). Redox potential is also important.

Despite the many complexities outlined above, some risk assessors try to model plant uptake of metals using simple regression equations (uptake *versus* partition coefficient K_d) that are intended to apply to all metals and all plant species. This highly generalised approach is used in the TOX-SCREEN model[8] and in the AERIS model.[9]

An obvious problem with this approach for generic risk assessment models is that the site-specific values for K_d are not available and one is forced to use default values taken from the literature. In practice, because of the difficulty of determining K_d experimentally, the same limitation usually affects site-specific risk models.

Given the complexities of plant–soil solution systems, and the widely different responses of different plant species, it is unlikely that any *general* regression approach is reliable for modelling plant uptake of metals. Unlike organic contaminants, however, it is feasible (and prudent) to examine the relevant plant uptake literature for each metal in isolation and to choose the best predictive equation (or equations) for that plant and metal. In principle, this approach would also allow consideration of factors such as competitive absorption.

For example Cd is preferentially displaced from adsorption sites when in competition with Pb and Cu[10] and hence its plant availability would be expected to increase. This corroborates the finding that soil Pb increases Cd uptake in plants.[11] Such effects are completely masked by multi-element regressions.

The difficulty with this approach is that for some metals there will be little or no data on uptake by relevant *garden vegetables*, and often important information about soil type, pH etc. is not reported.

Organic contaminants

A substantial literature exists on the uptake and translocation of organic chemicals used in agriculture, notably herbicides, pesticides and growth regulators. Many of these chemicals are fairly water soluble and some are weakly dissociating acids which undergo specific interactions

with plant tissues. Half-lives vary greatly but are often of the order of a few weeks. They thus have a quite different set of properties from the more lipophilic, nonpolar, stable chemicals that are of prime concern as industrial contaminants of human food sources. Correlations developed for agrochemicals should therefore be applied with extreme caution to these more hydrophobic chemicals.

There are four main pathways by which organic chemicals in soil can enter a plant, although most attention has been paid to root uptake and subsequent translocation by the transpiration stream. Most studies make no attempt to discriminate between this pathway and uptake via leaf pores of vapour in surrounding air.

One approach for estimating root uptake is to use empirical relationships between bioconcentration and octanol-water partition coefficient, K_{ow}. There are two obvious problems with this approach:

1. The regression coefficients are usually poorly constrained. For example the Travis and Arms regression[12] depends strongly on one point representing a compound group (polybrominated biphenyl) with a very large K_{ow} value. This means that the predicted behaviour of all organic compounds, including relatively water soluble ones, is strongly influenced by a single study of one exceptionally lipophilic compound.
2. It neglects work[13] that suggests that at lower values of log K_{ow} the bioconcentration factor *increases* with increasing log K_{ow}. On the other hand the Travis and Arms regression is based on a large quantity of experimental data and therefore has a *real-world* base that most theoretical models do not enjoy. In the absence of experimental data for relevant combinations of plant type, soil type and organic compound, uptake may need to be estimated using purely theoretical models. The fugacity type models pioneered by Paterson and Mackay[14] seem to hold most promise, but even they depend in part on correlations derived from experiments on particular plant species.

6.6 Transport via direct contact

Exposure to soil contaminants by direct contact takes place either by ingestion of soil (or of soil-contaminated plants), or by skin contact.

Direct ingestion of soil

It is widely appreciated that direct ingestion of contaminated soil, especially by young children, is an important exposure pathway. Three distinct categories of soil ingestion may be recognised:[15]

- inadvertent ingestion of soil by normal mouthing behaviour, and by accidentally dropping and picking up food items;
- occasional deliberate consumption of soil as part of normal exploratory behaviour in young children;
- geophagia (soil pica) – pathological consumption of soil, sometimes in relatively large quantities.

Early studies of soil ingestion by children were based on observations of mouthing behaviour coupled with measurements of dust or soil on children's hands.[16] More recent studies usually rely on mass balance of tracer elements. This approach is based on elements in soil that are not absorbed by passage through the body. Therefore, the amount of an element ingested from soil (and other sources) is taken as being the same as the amount excreted. When input sources other than soil and food are negligible, the input of an element from ingested soil can be calculated from the faecal output minus the input via food.

Unfortunately, there appears to be little agreement between the various tracer studies. Consequently, there is little consensus on the default values used by the various regulatory authorities for daily soil ingestion rate, and values for adult soil ingestion are little more than educated guesswork. This is an area that warrants much more research.

For soil contaminants that could enter the food chain via grazing animals, risk assessors should be aware that sheep and cattle can directly ingest substantial quantities of soil, especially when the forage cover is sparse. This is in addition to contaminant intake via consumption of grass or other vegetation.

Ingestion of soil attached to vegetables

Human exposure via consumption of home-grown vegetables needs to take account of surface contamination as well as contaminant uptake into plant tissue (discussed earlier). There is limited information on the soil load of vegetables. On the basis of a study carried out at Shipham, UK, which found high lead values in vegetables that left an insoluble residue after digestion, Alloway *et al.*[17] estimated that 0.36% of fresh weight was due to soil contamination. However, this was not confirmed for crop samples analysed for cadmium.

The Shipham result for lead is nevertheless close to that determined in an experimental study using beet leaves.[18] It is expected that some vegetables that are difficult to wash (e.g. leeks) or are eaten raw (e.g. lettuce) may carry a higher soil load than other vegetables. There is a distinct shortage of experimental studies, however.

Dermal exposure

Uptake of soil contaminants through skin contact depends on at least seven factors:

- area of skin in contact with soil;
- degree of soiling (mass per unit area);
- duration of contact;
- adherence of the contaminant to soil particles (which may vary with ageing), and partitioning into soil water and soil air, which will influence the rate of volatilisation;
- shielding effect (i.e. only a proportion of the contaminant is in direct contact with the skin);
- ability of the contaminant to penetrate the skin;
- contaminant concentration in soil and especially in the particle size fraction that adheres to skin.

Unfortunately, direct measurements of dermal uptake from soil are available for only a very small number of chemicals. In the absence of experimentally measured uptake data, predictive models are required (see Chapter 7). Particular care is needed in choosing models with a sound theoretical basis, and in validating them with experimental measurements whenever possible.

Skin contact with contaminants in indoor dust differs in several respects from exposure via outdoor soil. One factor is that only a fraction of house dust will be derived from outdoor soil. Also, indoor activity patterns, temperatures etc. are distinct from those outdoors. Thirdly, the adherence of dust may differ from that of soil because of differences in particle size, shape and moisture content.

6.7 Summary and conclusions

Contaminant hydrogeology

The transport and fate of contaminants in groundwater has become a major research area with a vast literature. Despite this underpinning, applications are still highly uncertain and rely on experience and intuition almost as much as on scientific knowledge. The paragraphs below indicate some of the areas where research needs to be focused if it is to help the practitioner succeed in delivering reliable risk assessments at an affordable cost.

Contaminant transport

Most research focuses on transport of contaminants in well-defined soils and groundwater aquifers. Most of the studies are carried out using homogenous sandy soils with high hydraulic conductivities. However, recent research has also pointed out the need to predict transport phenomena in much more heterogeneous soils and rocks.

Another common feature of the current research on contaminant transport is to focus on contaminants that are relatively easy to predict, such as highly soluble contaminants. But the transport and spreading of dissolved dense non-aqueous phase liquids (DNAPLs) such as the chlorinated aliphatics show a much more complicated pattern, and no good models exist for predicting the transport of these contaminants. The problem is even more challenging in cases with free product of these contaminants, which can form ganglia and smears along the flow path.

Chemical and physical processes

With respect to toxic metals, the importance of different processes and soil factors on partitioning and distribution in soils is currently being investigated by several soil, geochemical and environmental research groups. However, much of this research focuses on pure minerals and concentrations of toxic metals not typically found at contaminated sites. Most research has concentrated on the traditional toxic metals like cadmium, lead, copper, chromium and zinc, but it is evident that too little information is available on other potentially important trace metals and metalloids such as arsenic, antimony and vanadium. These contaminants are difficult to study and analyse, but they deserve much more attention with respect to partitioning and behaviour in the soil environment.

With respect to organic contaminants current research is mostly carried out under controlled conditions in terms of the environmental factors governing partitioning processes. Less research deals with more complex geological or geochemical situations. The spatial and time variability of the governing factors cannot, with current knowledge, be taken into account sufficiently in the prediction of contaminant transport in natural soils.

Biological processes

Most research on the fate of contaminants is carried out under controlled laboratory conditions, which makes it difficult to apply the results to natural soil environments. Such studies are not well suited for

addressing complex geological systems, and in most cases only one or a few contaminants are included in the study. The fate of complex mixtures of contaminants is seldom studied, and there is a strong need for *in situ* investigations of contaminant degradation.

Since degradability of organic contaminants is highly dependent on redox conditions, i.e. the availability of electron acceptors, it is important to pay attention to variations in redox conditions when carrying out biodegradation studies, including the spatial variability of this parameter.

Another research approach with limited relevance to the understanding of natural biodegradation processes and their kinetics is the use of newly added contaminants to soil in fate studies. Such an approach will not tell anything about the degradation of aged contaminants, which are probably much less bioavailable to the microorganisms and co-substrates necessary for degradation.

Modelling

There are different types of transport model, some describing the transport of contaminants in groundwater, some describing the air transport of volatile compounds, and a few dealing with more specific situations (see Chapter 7). It is important to distinguish between organic and inorganic compounds, since the different types of contaminant are affected by different processes. Most of the models developed to predict fate and transport of organic contaminants in soil and groundwater are based on the concept of advective transport and only include biodegradation, sorption and dispersion.

Data needs for transport and fate

The risk assessment practitioner needs practical guidance on what information is required to assess the significance of fate and transport processes at a specific site. Guidance is also required on how this information is to be interpreted. Information gathering for fate and transport analysis needs to be integrated into general site investigation protocols so that information is being gathered in a cost-effective manner. As site investigation will often be tiered, data needs for fate and transport analysis need to be considered at the various tiers at which site investigation might be conducted, ranging from preliminary to detailed investigations.

Many investigations are carried out without a clear understanding about the objectives. Some are designed to get a general picture of the distribution of contamination at a site; others aim at providing a good basis for scoping and designing remedial actions at the site; and some have the clear objective of providing a sound basis for risk assessment.

However, all these objectives are interrelated and it is highly recommended that investigations be planned to provide the necessary parameters for all purposes at the same time.

Exposure via inhalation and ingestion

Compared with contaminant hydrogeology, the scientific basis for predicting exposure via inhalation and ingestion is relatively undeveloped. There is a substantial literature on plant uptake of agricultural chemicals, but a rather sparse body of knowledge on uptake of industrial soil contaminants. There is a particular lack of knowledge with respect to most organic contaminants, and phytotoxic thresholds for most plant species are not well constrained.

Direct soil ingestion, especially by young children, is an important pathway but one that is difficult to investigate experimentally. There is a handful of studies based on soil tracer elements (e.g. elements such as Si and Al, which are not significantly absorbed by passage through the gut), but the experimental difficulties are so great that the results must be considered as having very low reliability.

Even if soil ingestion rates were reasonably well constrained, contaminant uptake from direct soil ingestion would still be difficult to predict. This is because the relative bioavailability of contaminants in soil is very poorly known.

Transport and fate of soil contaminants in air is relatively underresearched. Accurate prediction of human exposure via inhalation of fugitive dust is all but impossible using current modelling tools, and there have been few systematic attempts to measure fugitive dust emissions. Modelling capability for soil vapour transport, and intrusion into buildings, is better developed but there is a serious paucity of experimental data with which to corroborate model predictions.

With the current state of knowledge, it is inevitable that most transport and fate models are likely to over-predict exposure to soil contaminants. This is prudent and precautionary, but makes it very difficult to judge whether the costs of remedial action are proportional to the benefit.

References

1. Appelo, C.A.J. and Postma, D. (1993) *Geochemistry, Groundwater and Pollution.* Balkema, Rotterdam.

2. Domenico, P.A. and Schwartz, F.W. (1990) *Physical and Chemical Hydrogeology*. John Wiley and Sons, Chichester.
3. Chapelle, F. (1993) *Groundwater, Microbiology and Geochemistry*. John Wiley and Sons, Chichester.
4. Keenan, R.E., Saucer, M.M., Lawrence, F.H., Rand, E.R. and Crawford, D.W. (1989) Examination of potential risks from exposure to dioxin in sludge used to reclaim abandoned strip mines. In: *The Risk Assessment of Environmental Hazards: A Textbook of Case Studies* (D.J. Paustenbach, ed.). John Wiley & Sons, Chichester, pp. 935–998 (ISBN 0-471-84998-7).
5. Murphy, B.L., Toole, A.P. and Bergstrom, P.D. (1989) Health risk assessment for arsenic contaminated soil. *Environmental Geochemistry and Health* **11**, 163–169.
6. Franzen, D., Sackman, A., Oale, R. and Chopin, M. (1988) Analytical results report for ambient air and residential characterisation at Prospect Square, Park City, Utah. USEPA Hazardous Site Evaluation Division, Washington, DC
7. Steele, M.J., Beck, B.D., Murphy, B.L. and Strauss, H.S. (1990) Assessing the contribution from lead in mining wastes to blood lead. *Regulatory Toxicology and Pharmacology* **11**, 158–190.
8. Hetrick, D.M. and McDowell-Boyer, L.M. (1984) *User's Manual for TOX-SCREEN: Multimedia Screening-Level Program for Assessing the Potential Fate of Chemicals Released to the Environment*. Report EPA560/5-83-024. USEPA Office of Toxic Substances, Washington, DC.
9. AERIS (1991) *Aid for Evaluating the Redevelopment of Industrial Sites AERIS Model Version 3.0: Technical Manual*. AERIS Software Inc., Richmond Hill, Ontario.
10. Schmitt, H.W. and Sticher, H. (1986) Long-term trend analysis of heavy metal content and translocation in soils. *Geoderma*, **38**, 195–207.
11. Miller, J.E., Hassett, J.J. and Koeppe, D.E. (1977) Interactions of lead and cadmium on metal uptake and growth of corn plants. *J. Environ. Quality,* **6**, 18–20.
12. Travis, C.C. and Arms, A.D. (1988) Bioconcentration of organics in beef, milk, and vegetation. *Environmental Science Technology,* **22**, 271–274.
13. Briggs, G.G., Bromilow, R.H. and Evans, A.A. (1982) Relationships between lipophilicity and root uptake and translocation of non-ionised chemicals in barley. *Pesticide Science* **13**, 495–504.
14. Paterson, S. and Mackay, D. (1989) Modelling the uptake and distribution of organic chemicals in plants. In: *Intermedia Pollutant*

Transport: Modelling and Field Measurements (D.T. Allen, Y. Cohen and I.R. Kaplan, eds). Plenum Press, New York, pp. 269–282.
15. Ferguson, C.C., Krylov, V.V. and McGrath, P.T. (1995) Contamination of indoor air by toxic soil vapours: a screening risk assessment model. *Building and Environment* **30**, 375–383.
16. Duggan, M.J. and Williams, S. (1997) Lead-in-dust in city streets. *The Science of the Total Environment* **7**, 91–97.
17. Alloway, B.J., Thornton, I., Smart, G.A., Sherlock, J.C. and Quinn, M.J. (1988) Metal availability. In: The Shipham Report: An Investigation into Cadmium Contamination and its Implications for Human Health. *The Science of the Total Environment* **75**, 41–69.
18. Sheppard, S.C. and Evenden, W.G. (1992) Contaminant enrichment of sparingly soluble contaminants (U, Th and Pb) by erosion and by soil adhesion to plants and skin. *Environmental Geochemistry and Health* **14**, 121–131.

Chapter 7

Models

7.1 Introduction

Contaminated sites can be assessed either:

1. by comparing the measured levels of contamination with established guideline or screening values; or
2. by applying site-specific models whereby exposures and effects on receptors can be estimated for specific exposure scenarios.

Both procedures have distinctive roles in contaminated site risk assessment. Use of guideline or screening values is indispensable in preliminary assessment, and may be necessary to comply with legal requirements. Depending on their derivation, guideline values may also play an important role in site-specific risk assessment. However, screening values are typically based on *worst case* or *reasonable worst case* scenarios and consequently tend to be conservative. Hence, the role of site-specific models, particularly in detailed investigations, is to serve as an additional tool to minimise knowledge gaps and uncertainties in assessing risks. Models of both kinds, when applied appropriately, can improve the scientific reliability, transparency and clarity of a risk assessment and provide useful data for assessing cost effectiveness. Decision makers need to be aware that risk assessment of contaminated sites is highly multidisciplinary and may involve significant input from a wide range of scientific, medical and technical disciplines and from risk communication specialists.

Generally speaking, models are idealised and simplified representations of complex systems. In the context of contaminated land risk assessment, the word *model* is usually used in the following sense:

> A model (or suite of models for risk assessment) provides opportunities for drawing quantitative or semi-quantitative conclusions regarding release, transport, transformation and exposure with respect

to a measured or estimated contaminant concentration at a source (the contaminated site) and thereby allows assessment of the likely effects on receptors. Models suitable for such purposes should share a common set of scientific principles but be flexible enough to reflect the different conditions existing between different sites and among various countries and regions.

Key issues in designing contaminated land risk assessment models are summarised below.

- Scientific basis and methods. What mathematical relationships are used and why?
- Transparency in model application. What are the operating stages and how are they linked to each other?
- Transferability. What factors are important for site-specific application? Can local conditions and landuse scenarios be taken into account?
- Ability to upgrade models to reflect the latest scientific findings.
- Practicability and user friendliness. How easy is it to run the model for a specific input scenario? Can computer-based models be managed judiciously by persons without any data processing experience?
- Comprehensibility for non-experts. Can the results be understood by the various stakeholders? What graphic options are available to illustrate results?
- Ability to take account of variability as well as uncertainty. What uncertainties are included in the assessment, and how can natural variability of key parameters be taken into consideration?

7.2 Current practice

No comprehensive risk assessment model is available at present that would take into account all receptors of potential concern on contaminated sites. Instead, partial models and model components are in use related to specific receptors of concern. Risk assessment models may be classified into the following categories:

- Priority setting models for conducting relative assessments
- Risk assessment models for conducting quantitative or semi-quantitative assessment. These may include one or more of the following components:
  - human exposure models
  - toxicological models

– ecotoxicological models
– other transport models (e.g. dust and vapour transport; plant uptake)
– groundwater models (solute transport and fate).

Priority setting models

In risk assessment of contaminated sites, the characteristic feature of priority setting models is that they allow relative assessment for preliminary classification and decision making.

An early model, the Hazard Ranking System developed by the USEPA in the 1980s as part of the Superfund programme,[1] provided a relative assessment of (potentially) contaminated sites using a point system. The National Priority List (NPL) is based on this priority setting model. In Canada a national classification system (NCS) for contaminated sites has been developed[2] to provide a simple, consistent and reliable basis for classifying sites in terms of potential risks to human health and the environment. Using this system sites are placed into one of five categories: class 1, action required; class 2, action probably required; class 3, action may be required; class N, remedial action not required and class ?, insufficient data.

Because of the need for making consistent decisions covering large numbers of contaminated sites, most participating countries use priority setting models of one sort or another either at the national or local level. Priority setting models also exist for specific types of contaminated sites, for example the system developed by Gaz de France[3] for gasworks sites in France.

Some evaluation criteria commonly used to identify priority sites, or sub-areas for sampling and more detailed investigation, include:

- site type classification, e.g. hazardous waste site, abandoned industrial site;
- evidence for potentially contaminative uses;
- size of site;
- information on relief, slope and nature of soil and subsoil;
- estimation of soil permeability for air and water pathways;
- determination of distance to groundwater and aquifer sensitivity;
- current land use;
- existence of drinking water wells/service water wells/drinking water plant;
- distance to nature reserves or surface water bodies, and ground surface gradients.

A common feature of all priority setting models is that they give an initial weighting (often as a point score) with regard to the above criteria. This allows for qualitative comparisons between different sites.

Transport from soil models

Soil to groundwater transfer

Currently only a few models allow an assessment of pollutant transport in the unsaturated zone. This is due to the complexity and heterogeneity of processes occurring in the subsoil. To assess the risk of groundwater pollution, particular attention should be paid to pollutant transfer from soil to leachate and from leachate to groundwater. In the past, simple leachate balance models[4] were applied. Today, modern transport models provide a more rigorous approach. Most model calculations assume an equilibrium distribution of substances in a multiphase soil system. Some models also take biological/chemical degradation into account. Due to the complexity of the processes involved, many idealisations and simplifications are usually required. For example, the mobility potential of substances is attributed to a combination of a few physico-chemical parameters such as water solubility, kinematic viscosity, vapour pressure and organic carbon partition coefficient.

Soil to plant transfer

The transfer of pollutants from the soil into plants depends on contaminant properties, numerous soil properties (e.g. organic carbon content, complexing agents, pH-value, clay content) and on the response of the particular species or cultivar. Four approaches are commonly used to assess the transfer of pollutants into plant tissues:

- For some contaminants (heavy metals in particular) transfer factors are available from laboratory studies, some of which have been corroborated by random or systematic field tests.
- Other models are based on determining the bioavailability (the plant-available content of a contaminant), using a specific system of sample preparation and analysis e.g. ammonium nitrate digestion for metals. This is an area of active research.
- For most organic contaminants, the transfer from soil into plants is calculated using models such as the Briggs–Ryan model.[5,6] This is done on the basis of physico-chemical properties, mainly octanol-water partition coefficient and aqueous solubility. The basis of this type of transport model is the assumed equilibrium adjustment between soil solution/fine root system (RCF, root concentration factor)

and the resulting distribution up into the shoot (TSCF, transpiration stream concentration factor; SCF, stem concentration factor).

- For some organic contaminants, especially if they are chlorinated, the transfer from soil into plants is calculated using empirical regressions such as the Travis–Arms regression.[7] This calculates a bioconcentration factor from soil to plant on the basis of the octanol-water partition coefficient of the contaminant. The weakness of this approach is the large scatter of values around the regression (i.e. low correlation coefficient) and the sensitivity of the regression slope to one or two extreme values.

Soil to air transfer (vapour)

Release and transport models are required for volatile soil and groundwater contaminants, especially in the context of intrusion of toxic or explosive soil vapours into buildings. Some models include highly simplified diffusion-only algorithms (e.g. RBCA[8]); others include pressure-driven flow induced both by wind and temperature difference.[9–11] Many of the key parameters (soil permeability, air exchange rates in crawl spaces, floor leakage rates etc.) are difficult to determine experimentally. Soil vapour intrusion models are therefore very difficult to validate.

Soil to air transfer (suspended particles)

The suspended particles pathway is concerned with pollutant release and transport by wind. Influencing factors include wind velocity, soil structure and grain size, moisture content and vegetation cover. Dust particle size distribution is important because only fine dust (< 10 μ### m) can penetrate deep into the lung. Available models[12–15] appear to be highly conservative. For example, none appears to take proper account of soil moisture content in inhibiting the release of soil particles as dust.

Groundwater models

There are two categories of groundwater model in common use, those concerned with groundwater flow and those dealing with solute transport and fate in groundwater. None of the available models is capable of modelling the transport and fate of non-aqueous phase liquids (NAPLs) occurring as free product.

Geological and hydrogeological investigations are used to describe the fundamental geometrical and structural properties of an aquifer. Hydrogeological data (e.g. pumping tests) allow the quantification of flow-related parameters such as hydraulic conductivity and storage

capacity. To translate a hydrogeological conceptual model into a mathematical model, relevant physico-chemical laws (conservation law, flow laws, state equations) are applied and assumptions are made about continuity and hydraulic connectiveness. Since the late 1980s computer programs for modelling solute transport and fate have been widely used in connection with contaminated sites (e.g. MODFLOW, Aquifer Simulation Model (ASM), Hydrology Pre- and Postprocessor Groundwater Modelling System (HPP-GMS)). Such models tend to require substantial (and potentially expensive) input data. Hence simple screening-level models should also be employed so that more sophisticated models are only used on sites where they are really required.

Groundwater flow in fractured solid rock is usually modelled using so-called *dual porosity* models. Stochastic models have also been developed for modelling flow in fractured aquifers, although they are very difficult to validate.

Human exposure models

Human exposure models aim to quantify the transfer of site contaminants to humans (e.g. CSOIL,[13] UMS[14] and CLEA[15] in Europe, CALTOX in the USA and AERIS in Canada). The term *exposure* is defined here as intake of a contaminant via all relevant routes of entry into the human body (skin, respiratory tract, gastrointestinal tract). There is an important distinction between the amount that enters the body and the amount that is absorbed. The relative bioavailability of contaminants entering the gut or lung in a soil matrix is not well understood, and hence is difficult to model.

Exposure scenarios describe the groups of persons affected, including any characteristic behaviour and the relevant exposure pathways. This requires assumptions for which there is often limited experimental data. The consistency and reliability of inputs are thus of the utmost importance. Models and supporting documentation should conform to the following criteria:[16,17]

- assumptions should be plausible;
- uncertainties and variations in parameters should be clearly identified;
- the extent of scientific consensus should be discussed;
- relevant viable alternatives should be examined;
- gaps in knowledge should be clearly stated.

Most human exposure models are supported by extensive guidance relating to exposure parameters (e.g. in the Netherlands through CSOIL,[13] in Germany through the AGLMB[16] and the UMS,[14] and in

the UK through the CLEA model[15] and related procedural guidance). If experimentally justified databases do not exist, plausible assumptions about average or *high-end* conditions have to be made and justified. Unfavourable assumptions (*worst case*) have often been made in order to protect the population from the effect[5] of uncertainties. Again, if there are no empirically ascertained data, justified plausible assumptions about worst-case or reasonable worst-case conditions must be made.

This form of modelling (*point estimation*) can significantly overestimate exposure because the adverse assumptions in several model parameters accumulate. Therefore, the development of probabilistic assessment models is an active field of research. However, there are still problems, such as the choice of probability density function and the reliability of the procedure in the marginal sectors of the distributions. In the UK the CLEA model[15] is a probabilistic (Monte Carlo) model developed specifically for deriving soil guideline values for protecting human health. However, probabilistic models are somewhat controversial. For instance, in Germany they are not yet recommended for application.[16]

All exposure models are variants of the following simplified formula, with typical units shown for soil ingestion as an illustration:

$$E = A \times T \times R \times C/B$$

where E is exposure or absorbed dose (mg per kg body weight per day), A is soil intake rate (e.g. soil per day in grams), T is time of exposure (days), R is resorption rate (per day), C is concentration of contaminant in the uptake medium (mg per gram of soil) and B is body weight (kg).

This formula implies knowledge about contaminant concentrations in the uptake or contact media. If such information is lacking, transport models relevant to site conditions should be used (see Chapter 6). These models derive quantitative assessments of contaminant transfer (e.g. from soil into plants) but such assessments require a great deal of validation. The resorption rate (R) quantifies the transfer of contaminants from the intake medium into the systemic circulation on the basis of toxicokinetics (see Chapter 3). R is usually ascertained empirically and varies according to the exposure pathway. Some factors are age-dependent and relate to the gradual physiological and behavioural changes occurring during childhood and adolescence.

Receptor groups

For pragmatic reasons, receptor groups in many models are divided into children (say between 0 and 6 or 8 years) and adults (e.g. CSOIL[13] and

EPA[12]). Other models differentiate further between infants, small children, older children, adolescents and adults, e.g. UMS,[14,18] and CLEA.[15] Age-dependent parameters (e.g. body weight, skin surface area, breathing rate) are usually taken from national statistics. The data used to characterise human activity patterns and exposure are not currently satisfactory and further research is required.

The following exposure pathways are generally taken into account: ingestion (e.g. via soil, plants), inhalation (e.g. via dust particles, vapour) and percutaneous uptake (e.g. via soil). Which exposure pathways apply depends on the characteristics of the site and the contaminant.

Ingestion

Contaminant intake may occur through direct ingestion of soil or domestic dust. Relevant factors are:

- the quantity of soil or domestic dust ingested;
- the (measured) concentration in the medium;
- the accessibility of the soil or the degree of sealing;
- the availability and rate of resorption.

Resorption rates are substance-specific. They usually have their basis in human or animal studies but, for many contaminants, there are few data. *In vitro* methods for estimating resorption are an active area of current research, e.g. approximating the fraction available in the gastrointestinal tract using specific extraction techniques, for instance hydrochloric acid at pH 1–1.5 and 37°C.[19]

Indirect ingestion of pollutants can also occur by eating plants that have absorbed contaminants from the soil. Conservative assumptions regarding the proportion of those who grow their own fruit and vegetables currently tend to overestimate the risk for most of the population.

In addition, knowledge about the contaminant concentration in plants is required for calculating intake through fruit and vegetable consumption. As it can be difficult and costly to determine this information experimentally, it is often not available. Instead, plant uptake models which aim to quantify the transfer of contaminants from soil to plant are used (see Chapter 6.).

Inhalation uptake

Relevant factors in the investigation of inhalation uptake of gaseous or dustbound contaminants include:

- ventilation (breathing rate);

- contaminant concentration at the breathing zone;
- time of exposure;
- particle size distribution;
- deposition;
- resorption.

Ventilation rate depends on age and activity. There is a distinction between resting, and light, medium and intense physical exercise, with the daily ventilation rate consisting of a mixture of the activity-dependent breathing rates. An activity-related approach using time–budget studies[14] is suitable for measuring the peaks and troughs in breathing that occur during a day. For the assessment of breathing rate with continuous exposure a metabolic approach would be more suitable.[16]

The inhalation uptake from the respirable fraction (<10 μ###m) in airborne particles is generally quantified using substance-specific absorption coefficients. However, relevant data exist for just a few substances.

Percutaneous uptake

To assess potential percutaneous exposure, skin-, pathway- and substance-specific aspects must be considered. The following factors deserve particular emphasis:

- evaporation of volatile contaminants from the skin surface;
- reversible or irreversible bonding within the stratum corneum;
- metabolism or transport in the skin;
- anatomic region;
- state of hydration of the skin;
- influence of sweat on the skin surface.

It is necessary to make an assumption regarding the area of body surface that is actually exposed. Obviously this will depend on local weather conditions. It can generally be assumed that clothes and shoes are worn, and thus only certain parts of the body, such as hands, forearms, legs and face are likely to be exposed.

The adhesive capacity of the soil to the skin determines the contact with the contaminant and is soil-specific.[20] The absorbed quantity of contaminants from dermal contact with contaminated soil is further influenced by soil characteristics (grain size, soil organic matter, soil moisture, pH).

There is very little substance-specific information, e.g. from epidemiological studies or animal experiments, for quantifying dermal resorption. As a consequence, permeability coefficients are sometimes

modelled using chemico-physical parameters (e.g. EPA,[21] CLEA[15]). The transfer of pollutants is treated as a diffusion-controlled process, depending on the thickness of the soil layer, Henry's constant and the octanol-water partition coefficient. There are many uncertainties in modelling this uptake pathway.[22–24] In addition, few substances have adequate toxicological data for the dermal pathway.

Exposure scenarios

Normally different scenarios are examined (e.g. children's playgrounds, gardens/residential areas, parks and recreational areas, industrial areas, sports fields). Some computer-aided models allow site-specific situations and individual conditions to be considered e.g. RISC-HUMAN,[25] UMS,[14] and CLEA.[15]

Results of the exposure assessment

The final assessment can be performed in accordance with two principles:

- individual assessment of relevant exposure pathways, e.g. pathway-specific soil screening levels by EPA;[12]
- combined assessment of all parallel exposure pathways, in order to facilitate a comprehensive evaluation, e.g. UMS,[14] CSOIL,[13] and CLEA.[15]

In the presentation of results the contribution of the natural and/or anthropogenic background contamination may also be considered in order to indicate what proportion of the total exposure is derived from the investigated contaminated site.

The validity of an exposure model depends on the representativeness and reliability of measured values for direct contact media (e.g. soil, indoor air, domestic dust, drinking water, plants). If the results are based on transfer equations, with their inherent uncertainties, the results of the risk assessment should be qualified where necessary.

Toxicological models

Toxicity data used in contaminated land risk assessment models may be based on animal studies, occupational epidemiological studies, general-population epidemiology, *in vitro* tests on human cells etc. Most tolerable intakes derived for acute and chronic exposure are rather

poorly constrained and often make use of large, but rather arbitrary, uncertainty factors (see Chapter 3).

There is an increasing interest in using modelling techniques, particularly physiologically-based pharmacokinetic (PBPK) models, to help select the most appropriate animal studies and adapt them for use in the context of human exposure. There is also a large body of literature on statistical extrapolation of cancer bioassays to the low doses relevant to environmental exposure in humans. As far as we are aware, however, there are no toxicological models explicitly incorporated into contaminated land risk assessment models. Rather, whatever the modelling tools (if any) that were used to support toxicological studies, the results are simply incorporated into the tolerable daily intakes, cancer potency slopes etc. used in risk assessment models.

Ecotoxicological models

The role of ecotoxicology is to investigate the harmful impact of substances on biotic communities and to predict secondary effects on the aquatic and terrestrial ecosystems, populations and species (see Chapter 4). Obviously, where a whole ecosystem is the *receptor* the problems associated with estimating ecological effects are enormous. Moreover, effects such as that on the interactions within and between populations are often difficult to measure. Therefore in many cases the ecotoxicological models developed are only partial, and are primarily for aquatic ecosystems. However, toxic effects on soil ecosystems play an important part in deriving Dutch intervention values.[26]

Ecotoxicological evaluation methods, and existing approaches for qualitative assessment of adverse influences on ecosystems, are relatively well developed (e.g. in the USA[27] and Canada[28]). Here the principles have largely been adopted from other disciplines, in particular from regulations for the approval of chemical substances, e.g. OECD,[29] EPA,[30] van Leeuwen and Hermans.[31] These regulations use different tests, including ecotoxicological tests or *bioassays* to rate the ecological compatibility of chemical substances. They are not explicitly derived for an assessment of contaminated sites. In detail of number and selection of species and the methodological procedures, bioassays and field studies are not standardised, but the following test strategies in particular warrant special mention:[29,30,32,33]

- degradation behaviour in the soil (e.g. determining the rate of mineralisation);
- mobility in the soil (*in situ*, laboratory);

- plant test (monokotyle, e.g. *Avena sativa*; dikotyle, e.g. *Brassica rapa*);
- soil microflora (enzyme tests, respiratory tests, nitrogen conversion, layer degradation);
- earthworm test (e.g. Eisenia, Lumbricus);
- collembola test (e.g. *Folsomia candida*);
- carabidae test (e.g. *Poecillus cupreus*).

To extend this approach synthesised test methods – model ecosystems in particular – have been developed but they are generally related to individual cases. Some representative examples are the plant metabolism box (NATEC[34]), the model ecosystem of the BBA, Berlin,[35] and the TME soil columns (*Microcosm*) of the EPA.[36] Predicted concentrations of no environmental concern are derived in connection with contaminated sites from data on biological degradation, mortality (LD50/LC50) and impact doses (ED50/EC50 as well as NOEC and LOEC) acquired in bioassays. The derivation usually involves extrapolation of bioassay results and application of safety factors. Relevant soil data are available for practical applications, e.g. the Netherlands (maximum acceptable levels), Denmark (ecotoxicological quality criteria) and Germany (concentration of no environmental concern). However, the last is not in use at present.

Uncertainties and probabilistic approaches

Inaccuracies and uncertainties are naturally associated with modelling and are further increased by the natural variability of parameters. The most important aspects include:

- uncertainties resulting from model limitations (e.g. aspects neglected in the abstraction process);
- uncertainties arising from set conventions (e.g. observation period);
- uncertainties where validation data are lacking (e.g. dermal absorption from contaminated soil);
- uncertainties in the scientific basis (e.g. in the toxicological database);
- uncertainties associated with site data (e.g. due to sampling design, analysis);
- variability of exposure factors (e.g. statistical distribution of uptake rates and exposure periods);
- variability in the influencing physico-chemical factors and their influence on transport and fate models.

All model variables are subject to stochastic variation. Therefore, realistic exposure assessments should take into account the *distribution* of exposure factors (e.g. user physiology, user behaviour, frequency of exposure, periods of exposure). At present this is not general practice. Among the available probabilistic approaches Monte Carlo analysis is of particular note.[15,17] If probability density functions replace single-valued input parameters in exposure models, the probability of exposure occurring at certain levels can be estimated following repeated modelling from the distribution of output results.

There is some debate about whether probabilistic approaches should also be used for the derivation of human acceptable exposure limits, e.g. tolerable daily intake, and in some countries (e.g. the Netherlands) this is beginning to be accepted.[37]

Of the models discussed earlier, only the CLEA model was designed as a probabilistic model. A number of commercially available models are probabilistic in nature or can be adapted for such use by linking with other models (e.g. CrystalBall). The main problem with all such models is finding sufficient reliable data to constrain the input distributions. It is particularly important to recognise, and correctly deal with, input variables that are correlated.

7.3 Summary and conclusions

The various types of model and model-related issues are briefly summarised below. With rapid growth in the availability of models for assessing risks from contaminated sites, it will become increasingly difficult for regulators and other stakeholders to understand the detailed scientific basis of the models and the many assumptions underpinning them. For this reason, among others, it seems likely that many countries will prefer to focus attention on a small number of models that have been thoroughly peer reviewed for use within the national context.

- **Priority setting models**. There is widespread use of various priority setting models in the participating countries, both at the local and national level.
- **Transport models**. Models for the transport of contaminants are in various stages of development. Groundwater models of varying degrees of complexity have been developed and are now routinely used in risk assessment in many countries. In contrast, only the Netherlands and the UK in Europe have produced comprehensive soil to indoor air transfer models. There is still an immense lack of

knowledge about contaminant behaviour in various media under non-equilibrium conditions.

- **Exposure models**. Relatively comprehensive exposure models have been developed in the Netherlands, Germany, the UK, the USA and Canada. They all assess exposure on the basis of the source–pathway–receptor paradigm.
- **Model application**. Models may be primarily applied within detailed risk assessment, or to derive screening or guideline values (see Chapter 8). Computer-based risk assessment models are still not widely used in most participating countries although this appears to be changing rapidly as available models become more user friendly.
- **Variability**. Variability and uncertainty are recurrent themes in risk assessment, of particular significance when considering the strengths and limitations of modelling.
- **Ecosystem models**. Assessing the risks to ecosystems differs widely between participating countries and the USA and Canada. In Canada and the Netherlands risks to ecosystems are assessed using a model with a clearly defined methodical basis, whereas in most participating countries such models are only in the early stages of development.
- **Importance of models**. The application of screening and guideline values with a firm scientific basis to the assessment of contaminated sites is not in contradiction with the use of risk assessment models. Even countries with advanced risk assessment models use guideline values because they are important in helping to achieve consistency and cost effectiveness, and increasingly are based on sound risk assessment principles. More detailed modelling can then be reserved for sites where the extra costs can be justified.
- **Model optimisation**. Risk assessment models are only plausible and comprehensible if they have a sound scientific basis. Thus, when there are experimentally determined data that can be realistically projected outside the laboratory context they should be used for model validation.
- **Dealing with inaccuracies and uncertainties**. Models inevitably contain inaccuracies and uncertainties, which are magnified by the variability of the input parameters. The development of systematic methods for addressing this problem would be very beneficial.
- **Risk- and economic-based models**. There is currently no comprehensive model that fully integrates all aspects of risk assessment. In particular, there is an absence of models that include the effects of risk perception. Likewise, there appear to be no generally available models that incorporate economic considerations. Comprehensive

assessment of the benefits of remedial action requires quantified evaluations of all reasonable alternatives. The use of cost–benefit analysis will increase in importance in the future but detailed discussion of its possibilities and limitations is beyond the scope of this book.

References

1. EPA (1988) *Superfund Exposure Assessment Manual*. Environmental Protection Agency, Washington, DC.
2. CCME (1996) *Guidance Manual for Developing Site Specific Soil Remediation Objectives for Contaminated Sites*. Canadian Council of Ministers of the Environment.
3. Costes, J.M., Texier, J., Zmiron, D. and Lambrozu, J. (1995) A priorization system for former gaswork sites based on the sensitivity of the environment. In *Contaminated Soil* '95 (W.J. van den Brink, R. Bosman and F. Arendt, eds). Kluwer, Dordrecht, pp. 605–607.
4. Schroeder, P.R. *et al.* (1984) *The Hydrologic Evaluation of Landfill Performance (HELP) Model, Handbook and Documentation*. Environmental Protection Agency, Washington, DC.
5. Briggs, G.G., Bromilow, R.H., Evans, A.A. and Williams, M. (1985) Relationships between lipophilicity and the distribution of non-ionised chemicals in barley shoots following uptake by the roots. *Pesticide Science* **14**, 492–500.
6. Ryan, J.A., Bell, R.M., Davidson, J.M. and O'Connor, G.A. (1988) Plant uptake of non-ionic organic chemicals from soil. *Chemosphere* **17**, 2299–2323.
7. Travis, C.C. and Arms, A.D. (1988) Bioconcentration of organics in beef, milk and vegetation. *Environmental Science and Technology* **22**, 271–274.
8. American Society for Testing and Materials (1995) *Emergency Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites*. ASTM E-1739. ASTM, Philadelphia, PA.
9. Waitz, M.F.W., Freijer, J.T., Kreule, P. and Swartjes, F.A. (1996) *The VOLASOIL Risk Assessment Model Based on Csoil for Soils Contaminated with Volatile Compounds*. Report no. 715 810 014. National Institute of Public Health and the Environment (RIVM), The Netherlands.
10. Ferguson, C.C., Krylov, V.V. and McGrath, P.T. (1995) Contaminations of indoor air by toxic soil vapours: a screening risk assessment model. *Building and Environment* **30**, 375–383.

11. Krylov, V.V. and Ferguson, C.C. (in press) Contamination of indoor air by toxic soil vapours: the effects of subfloor ventilation and other protective measures. *Building and Environment* 00, 000–000.
12. EPA (1996) *Soil Screening Guidance: Technical Background Document*. Environmental Protection Agency, Washington, DC.
13. Van den Berg, R. (1991/1994) *Human Exposure to Soil Contamination: A Qualitative and Quantitative Analysis Towards Proposals for Human Toxicological Intervention Values*. Report no. 725 201 011. National Institute of Public Health and the Environment (RIVM), The Netherlands.
14. Hempfling, R., Doetsch, P., Stubenrauch, S., Mahr, A., Bauer, O., Koschmieder, H.J. and Grünhoff, D. (1997) UMS-System zur Altlastenbeurteilung – Instrumente für die pfadübergreifende Abschätzung und Beurteilung von altlastenverdächtigen Flächen, Abschlußbericht F+E-Vorhaben 109 01 215, Umweltbundesamt (UBA), Berlin.
15. UK Department of the Environment, Transport and the Regions (forthcoming) *The Contaminated Land Exposure Assessment Model (CLEA): Technical Basis and Algorithms.* CLR report no. 10. DETR, London.
16. AGLMB (Arbeitsgemeinschaft der leitenden Medizinalbeamtinnen und Beamten der Länder (1995) Standards zur Expositionsabschätzung, Bericht des Ausschusses für Umwelthygiene (Behörde für Arbeit Gesundheit und Soziales, Hrsg.), Hamburg.
17. EPA (1997) *Policy for Use of Probabilistic Analysis in Risk Assessment, 15.05.1997, Guiding Principles for Monte Carlo Analysis in Risk-Assessment*, EPA/630/R-97/001. http:// www.epa.gov/-ncea/mcpolicy.htm
18. Wichmann, H.E., Ihme, W. and Mekel, O. (1993) Quantitative Expositions- und Risikoabschätzung für drei kanzerogene Stoffe in Altlasten, GSF-Bericht.
19. Umweltministerium Baden-Württemberg, Germany (1993) Verwaltungsvorschrift des Umweltministeriums zum Bodenschutzgesetz über die Ermittlung und Einstufung von Gehalten anorganischer Stoffe im Boden, Az. 44-8810.30-1/46, 24 August.
20. Finley, B., Scott, P. and Mayhall, D. (1994) Development of a standard soil-to skin adherence probability density function for use in Monte Carlo analyses of dermal exposure. *Risk Analysis* **14**, 555–569.
21. EPA (1992) *Dermal Exposure Assessment: Principles and Applications*, interim report. Environmental Protection Agency, Washington, DC.

22. McKone, T.E. (1990) Dermal uptake of organic chemicals from a soil matrix. *Risk Analysis* **10**(3), 407ff.
23. McKone, T. and Daniels, J. (1991) Estimation human exposure through multiple pathways from air, water and soil. *Regulatory Toxicology and Pharmacology* **13**, 36–61.
24. Kissel, J., Richter, K. and Fenske, R. (1996) Field measurement of dermal soil loading attributable to various activities: implications for exposure assessment. *Risk Analysis* **16**, 115–125.
25. Goldsborough, D.G. (1995) Risc-Human: a computer model for calculating site specific human exposure. In: *Contaminated Soil '95* (W.J. van den Brink, R. Bosman and F. Arendt, eds). pp. 613–614.
26. van den Berg, R., Denneman, C.A.J. and Roels, J.M. (1993) Risk assessment of contaminated soil: proposals for adjusted, toxicologically based Dutch soil clean-up criteria. In: *Contaminated Soil '93* (F. Arendt *et al*., eds). Kluwer, Dordrecht, pp. 349–364.
27. EPA (1989) *Risk Assessment Guidance for Superfund*, vol. II: *Environmental Evolution Manual and Ecological Assessment of Hazardous Waste Sites – Field and Laboratory Reference*. Environmental Protection Agency, Washington, DC.
28. CCME (1996) *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines*. Canadian Council of Ministers of the Environment.
29. OECD (1984) *Guideline for Testing of Chemicals*, nos. 207 and 208. Organization for Economic Cooperation and Development, Paris.
30. EPA (1982) *Pesticide Assessment Guidelines Standards*, subdivision N, sections 162, 163 and 165. Environmental Protection Agency, Washington, DC (cited in reference 29).
31. Van Leeuwen, C.J. and Hermens, J.L.M. (1996) *Risk Assessment of Chemicals: An Introduction*. Kluwer, Dordrecht.
32. OECD (1981) *Guideline for Testing of Chemicals*, no. 304A, *Inherent Biodegradebility in Soil*. Organization for Economic Cooperation and Development, Paris.
33. Battelle Europe (1993) *Basis for the Assessment of the Ecotoxicological Potential of 'Old Chemicals' in the Terrestrial Environment – Development of a Testing Strategy* (English summary), F + E Vorhaben, Umweltbundesamt (UBA), Berlin no. 106 04 103.
34. Figge, K., Klahn, J. and Kock, I. (1993) Testing of chemicals by evaluation of their distribution and degradation patterns in an environmental standard system. *Regulat. Pharm.* **3**, 199–215.
35. Schuphan, I., Schärer, M., Heise, M. and Ebing, W. (1987) Use of laboratory model ecosystems to evaluate quantitatively the behaviour of chemicals. In: *Pesticide Science and Biotechnologies* (R.

Greenhalgh and T. Robub, eds). Blackwell Science, Oxford, pp. 437–444.
36. EPA (1987) Soil core microcosm test 797.3995. *Federal Register (USA)* **52** (187).
37. Slob, W. and Pieters, M.N. (1997) *A Probabilistic Approach for Deriving Acceptable Human Intake Limits and Human Health Risks from Toxicological Studies: General Framework*. Report no. 620 110 005. National Institute of Public Health and the Environment (RIVM), The Netherlands.

Chapter 8

Screening and guideline values

8.1 Introduction

Most industrialised countries are currently drawing up or revising policies and procedures to deal with contaminated land. To implement these policies effective tools must be developed to assess the need for remediation and to make decisions about remediation strategy. Risk assessment is, if properly performed, an objective way of assessing the potential impact of soil pollution on human health, ecosystems and the environment in general. Risk assessment may involve a very detailed investigation of all sources, pathways and receptors of concern at a given site. This may be a lengthy and costly process. Therefore, a tiered approach to assessing suspect sites is sensible. Expensive risk assessments are restricted to those sites that are likely to pose significant risks *and* where decisions about remediation are difficult due to the complexity of the site and/or the costs of remediation. Many countries advocate or endorse the use of screening values, trigger values or guideline values as a component in risk assessment, or to facilitate *rule of thumb* decisions about the need for remediation at smaller sites where a more detailed risk assessment would lead to disproportionate costs.

Bearing in mind the generally accepted idea that risk assessment is a key element in risk management, many countries favour appropriate use of a generic approach in which measured pollution concentrations are compared with risk-based guideline values, thus allowing rapid and consistent risk assessment of contaminated sites. The success of such an approach depends on the following:

- the scientific basis used for deriving the guideline values;
- the feasibility of developing generic landuse scenarios;
- supporting advice on the role of guideline values within the contaminated land assessment procedure, and the implications of exceeding the values.

It is nevertheless valid to ask the question: to what extent are screening and guideline values useful tools in the decision making process for contaminated site management? For many years three issues have been central to the debate surrounding the use of soil screening and guideline values: the reliability of the values calculated; the treatment of uncertainty; and the relationship between generic scenarios and real site conditions. Siegrist[1] was one of the first authors to discuss the advantages and disadvantages of using generic values in risk assessment of contaminated sites. These are listed below although, it should be noted that some of the disadvantages apply equally to site-specific risk assessment techniques.

Advantages

- speed and ease of implementation;
- similar sites would be handled in a similar way;
- useful for initial assessment of significance of contamination;
- *a priori* information facilitates planning and action;
- encourages developers to undertake decontamination/restoration;
- potential consistency with strategies for environmental standards;
- reality of contaminated land made easy for the layperson;
- facilitates environmental audits of industrial sites;
- facilitates monitoring/permitting of operating industrial sites;
- can be used for performance assessment of soil treatment plants;
- implies non-negotiability and reduces local political influences.

Disadvantages

- some key site-specific considerations may not be accounted for;
- standards, guidelines and criteria are not formulated for many toxic substances of concern. Existing guidelines formulated under other programmes are not necessarily appropriate for contaminated land;
- lack of toxicological criteria for many substances;
- generic values imply a level of understanding, knowledge and confidence which may not exist;
- once values are established, site-specific flexibility may be difficult.

The list of advantages and disadvantages presented by Siegrist still has some relevance today although it does not reflect more recent thinking on the role of generic values in risk assessment. In particular, it is useful to make a distinction between *screening* values, which are typically used in preliminary risk assessment, and *guideline* values (of varying degrees of sophistication) which are used more widely in risk assessment. The comments below are intended to clarify these concepts, although the distinction between the two is not always sharp.

Screening values are generic values intended to screen out those sites (or parts of sites) for which risks are considered too small to warrant more detailed investigation. They tend to be based on very pessimistic exposure assumptions and/or very stringent criteria for maximum tolerable risk.

Guideline values are designed to provide generic guidance to risk assessors on the significance of contaminant concentrations in soil, groundwater or other media. At one end of the spectrum they may be based on rigorous multi-pathway probabilistic risk analysis of generic exposure scenarios. At the other extreme they include, as a limiting case, the most basic of screening values.

The important point, however, is that the debate has moved away from the merits and demerits of *generic* and *site-specific* risk assessment. It is now widely recognised that screening and guideline values are tools to be used, when appropriate, in site-specific risk assessment (indeed, a risk assessment that is not site-specific is not a risk assessment at all). This places the onus on the risk assessor to understand the derivation of screening/guideline values, and their limitations, in order to be able to judge when they can be used with confidence.

If properly used, screening and guideline values have an important role to play in contaminated land policies. They provide a relatively simple method for assessment of risks, which is comprehensible both for professionals with backgrounds in various fields and for the general public. Guideline values facilitate the management of contaminated sites at the administrative level and, moreover, can lead to important savings of money and time. Finally, a system of values helps to establish a comparable and meaningful scale for different pollutants, which may be important for soil protection policies.

An example of the historical controversy surrounding the use of generic values is the late establishment (1996) of soil screening levels in the USA.[2] In contrast, UK guidelines date back to 1979,[3] and the Netherlands has had legal values since 1983.[4] Today, screening/ guideline values (S/G values) are widely accepted as valuable tools in risk assessment if properly used. The reasons for this change in viewpoint may be related to:

- a clearer definition of the values and their role within the decision making process; and
- increasing confidence on the part of governments and regulators with respect to the appropriateness of the values, and improvements in the scientific basis underlying their derivation.

Although the advantages of generic values are now generally recognised, there are important differences between countries with regard to their role and definition. These are reviewed and analysed below. After that, the current practice in the derivation of the values is addressed. This is centred on the question of whether guideline values fulfil the role for which they are designed.

8.2 Different roles of screening/guideline values

As in other areas of environmental protection (air, surface waters, coastal water, etc.), two different branches with different aims are recognised within soil protection policies: the prevention of new contamination and the remediation of already contaminated land. In the following subsection, the different types of generic value that may be applied in each case are reviewed, although emphasis is given to those values related to the assessment of contaminated sites.

(a) Values applied in the prevention of new soil contamination

Two types of values may be considered:

1. Emission limit values for pollution sources (air emissions, leachate release, etc.). These limits are based on the potential for dispersion and on the toxicity of a substance, with the generic aim of protecting the quality of the medium receiving the released substance. Generally, emission limits are not specifically mentioned in connection with soil protection laws, and more often are included in regulatory documents applying to specific activities and/or in the context of general environmental protection laws. The most relevant examples are the regulation of sewage sludge application to agricultural soils, and regulations concerning the re-use of materials (e.g. building construction waste and decontaminated soil materials).
2. Values for the receiving media, e.g. soil and groundwater. These are not always considered within regulatory documents concerning soil protection, although some exceptions can be found. These values are established according to:
   - background concentrations which would represent the lowest reasonably achievable limit, or
   - higher than background concentrations which do not pose an unacceptable risk even considering the most sensitive landuse and receptors.

Depending on their derivation both types of value represent a negligible risk of adverse effects on the soil or on other receptors through soil exposure. They may be used as reference levels which, if exceeded, indicate the need for further assessment, monitoring or perhaps other measures.

(b) Values applied in the management of already contaminated sites

According to the different phases in the assessment and management of contaminated sites, the following types of value may be relevant:

1. Values used in prioritization of potentially contaminated sites. Some countries maintain an inventory of suspected contaminated sites, usually based on historical surveys of landuse, which can be categorised using a prioritisation system, either qualitative or quantitative. Such systems may be used for initial assessment of contaminated sites, based on sensitivity of the landuses considered; but more often, they represent a classification system by means of which sites are scored according to relative hazard (based, for example, on a contingency/impact matrix) and perhaps on economic factors.
2. Values used in the site investigation and decision making process. Most existing guideline values belong to this category. Depending on their meaning or definition, these S/G values may be classified as follows:
   - *Background values* (R0): these establish a reference level corresponding to unpolluted or non-anthropogenic conditions. It is usually assumed that they constitute a negligible risk.
   - *Values corresponding to the upper limit of acceptable risk in a conservative derivation* (R1): these represent a level below which the resulting risk is considered acceptable, and it may be negligibly small. Above this value significant risks are more likely and consideration of site-specific conditions and a more detailed assessment may be called for.
   - *Values corresponding to the upper limit of acceptable risk in a realistic derivation* (R2): if soil concentrations are above these values there is a higher degree of presumption about the existence of unacceptable risks.
   - *Values corresponding to the upper limit of acceptable risk in a realistic derivation, and exposure scenarios that are not affected by site-specific variability* (R3): this includes generic values derived on the basis of acute risks to sensitive receptors. The presumption

is made that contaminant concentrations in excess of these values *will* result in unacceptable harm or damage.

These different definitions depend on the degree of certainty about scenario characterisation, and on the approach used in deriving the generic values. Both factors influence the role these values play within the risk assessment protocols adopted. The following functions may be distinguished:

- to differentiate between natural and anthropogenic concentrations. This is the role of R0-type values;
- to establish the need for further investigation and more detailed assessment. Values of type R1 and R2 have this kind of function;
- to establish the need for remediation. Values of type R2 or R3 are more commonly used with this aim;
- to establish remediation objectives. In some countries S/G values are used as remediation targets, when the remedial technology is available at a reasonable cost. Two kind of values are found:
  - remediation targets constituting negligible risk; these are R0-type values;
  - remediation targets based on risk acceptability criteria related to landuse. These may be identical to R1 or R2 type values, but are not necessarily so. Indeed, it is more usual for remediation objectives to be drawn up on a case by case basis.

A summary of the approaches adopted by different countries, together with the types of value used, is presented in the following section.

8.3 Current practice

Approaches in the use of screening/guideline values

In general terms, three different approaches can be distinguished in connection with the relative role that S/G values play in the site assessment process.

Type A

Guideline values represent a limit above which intervention will be required; they indicate the need for remediation or other measures. However, it is more detailed site-specific assessment which establishes the urgency of action. This approach is followed by Denmark, the Netherlands and Italy.

Type B

Screening values constitute a screening tool indicating the need for further investigation and/or the need to carry out a more detailed site-specific risk assessment. Austria, Flanders, Finland, France, Germany, Norway, the Basque Country, and Switzerland use Type B screening values as a component in simplified risk assessment. The key assumption in this approach is that current methodologies, or the kind of landuse, do not permit the derivation of generic values that accurately reflect site circumstances. Nevertheless, because of the conservative approach used to derive the values, they can be used successfully to assert the absence of unacceptable risks.

Type C

Use of guideline values is encouraged but is optional. Depending on the characteristics of the site investigated, and especially its similarity to the generic landuse scenario, the site investigator must determine the appropriateness of using the values in risk assessment. Should their application be deemed appropriate, the amount of money and time saved in the assessment process is a clear advantage. This approach is followed by Greece, Portugal, Sweden and the United Kingdom.

A schematic representation of the relationships between types of generic value and types of country approach is given in Figure 8.1. In practice there are subtle gradations along both axes, and it is unlikely that the conditions in any country will exactly correspond to the nodes on the diagram.

It may be concluded that, if S/G values are derived using the same scientific basis and technical tools used in site-specific risk assessment, the major difference between the two approaches relates to how closely the generic exposure assumptions used in deriving the S/G values correspond to exposure conditions at a site. The reluctance of some countries to embrace the S/G value approach seems to be related mainly to its reputation for being excessively conservative. There is a strong drive in several countries towards developing S/G values that are both realistic in their exposure assumptions and protective of the environment and human health.

Differences between countries

The countries reviewed in compiling this section are the 16 participating countries plus Australia, Canada and the USA. Screening values used by regional governments have also been reviewed, although

detailed assessment has focused on values established at the national level, except for those countries where national values have not been adopted.

As shown in Table 8.1, all countries included in the assessment, with the exception of Ireland, are using or intending to use S/G values in the context of their policies on contaminated land. Some countries are making temporary use of foreign values, although most countries have derived their own values or are doing so. The S/G values that have been drawn up sometimes refer to the soil medium; but in other countries they also refer to groundwater and in a few cases to surface water and soil air as well.

Given that the scientific issues underpinning the derivation of S/G values are discussed in more detail in other chapters, and from a general methodological point of view in Chapter 2, the main aim of this analysis has been to understand the differences between countries, with respect to the aspects considered in the following subsections.

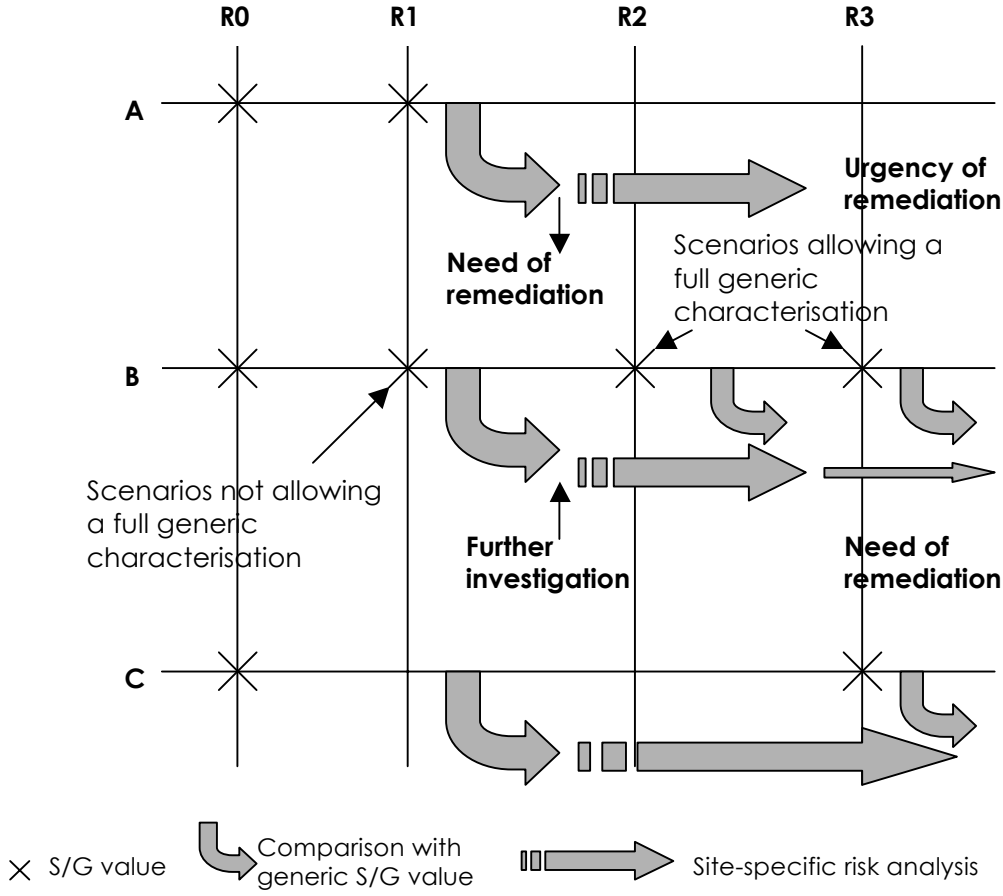

Figure 8.1 Graphical representation of generic value types (R0, R1, R2, R3) and approaches taken in different countries. See text for discussion

Table 8.1 S/G values within the soil policy framework in different countries

Country	S/G values	Nomenclature	Comments
Austria (A)	Yes	Screening values (a,e)	Published
		Action values (a,e)	Published
Belgium (B)			
Flanders	Yes	Remediation values (s,g)	Published/in prep.
	Yes	Background values (s,g)	Published
Wallonia	Intended		
Brussels	Intended		
Denmark (DK)	Yes	Quality Criteria (s,g)	Published
		Background values (s,g)	Published
Finland (FIN)	Yes	Target values (s,g)	Dutch values
		Limit values	In prep.
		Background values (s,g)	Published
France (FR)	Yes	VCI (Impact Assessment Value) (s,g)	In prep.
		VDS (Source Definition Value)	In prep.
Germany (D)	Yes	Trigger values (s)	Published/in prep.
		Action values (s)	Published/in prep.
		Background values (s,g)	Published
Baden-Württemberg	Yes	Background & assessment values (H-P) (s)	Published
Bavary	Yes	Soil values (SV) I, II, III (s)	Published
Berlin	Yes	Action & Reutilisation values	Published
Hambourg	Yes	Remediation values (s,g)	Published
Nordrhein-Westfalen	Yes	Limit values (s)	Published
Saxony	Yes	Assessment & clean-up values (s)	Published
Greece (GR)	Yes	Water Guideline values (w)	Int. stand. too
Ireland (IR)	Not int.		
Italy (IT)	Yes	Acceptable levels	In prep.
Piemonte	Yes	Acceptable/Remediation Levels	Int. stand.
Toscana	Yes	Acceptable/Remediation Levels	Int. stand.
Lombardia	Yes	Acceptable/Remediation Levels	Int. stand.
Emilia Romagna	Yes	Acceptable/Remediation Levels	Int. stand.
Netherlands (NL)	Yes	Intervention values (s,g)	Published
		Target values (s,g)	Published
Norway (N)	Yes	Threshold values (s)	Danish/Dutch values
Portugal (P)	Yes	Remediation values	Canadian values
Spain (SP)	Intended		
Basque Country	Yes	B-assessment values (s)	Published/in prep.
		C-assessment values (s)	Published/in prep.
		A-assessment values (s)	Published
Sweden (SW)	Yes	Generic soil guideline values (s)	Published
Switzerland (CH)	Yes	Trigger values (s)	Published/in prep.
		Clean-up values (s,e,a)	Published/in prep.
		Guideline values (s)	Published/in prep.
United Kingdom (UK)	Yes	Guideline values (s)	In prep.
		Trigger threshold values (old) (s)	Published
		Trigger action values (old) (s)	Published
Australia (AUS)	Yes	Investigation levels (s)	
		Background levels	
Canada (CAN) CCME	Yes	Assessment values (s,w/g)	Published
		Remediation values (s,w/g)	Published
Alberta	Yes	Tier I criteria (s)	Published
British Columbia	Yes	Reference and action values (s,w/g)	Published
Ontario	Yes	Clean-up criteria (s)	Published
Quebec	Yes	A,B,C values (s,g)	Published
USA	Yes	Soil Screening Levels (s)	Published
		MCLS (w/g)	Published

s: soil; g: groundwater; w: surface water; a: soil air; e: eluate

Derivation of values aimed at protecting human health

- All countries use similar sources of toxicological information (see Chapter 3). WHO and USEPA data compilations are generally preferred, with the recommendations being reviewed and revised as appropriate by national panels of experts. The main problem in the use of these reference doses arises from the fact that most relate to pollutant bioavailability in food additives or drinking water, and do not take into account the influence of the soil matrix.
- Most of the factors involved in exposure assessment (exposure duration and frequency, averaging time, etc.) are defined by activity patterns and characteristics of the most sensitive receptors, and may therefore vary from country to country. However, the basic information and models used to determine the final intake rate by humans are clearly similar; this is a field where collaboration between countries should be possible and very useful. In this sense, it seems reasonable to focus efforts on the characterisation of key parameters and exposure pathways for the different landuse scenarios.
- There are differences between countries with regard to the criteria adopted for risk acceptability. For non-carcinogenic compounds, soil allocation factors applied in the calculus of S/G values vary from 10% to 100%, whereas for carcinogenic compounds tolerable risk levels vary from 10^{-6} to 10^{-4} (theoretical excess lifetime cancer risk).

Derivation of values aimed at protecting ecosystems or other ecological receptors

Only five countries have derived values for the protection of ecological receptors, and the procedures used are very similar. Basically, the following features are typical (see also Chapter 4):

- Use of NOEC, LOEC or LC50 values obtained in laboratory tests together with adequate safety factors for toxicity assessment for the different organisms.
- In most cases, the assessment only considers exposure through direct contact of soil-dwelling species with polluted soil. Exposure through the food chain for animals living on the surface may be considered for specific contaminants. However, the required information on ingestion rates, and pollutant concentrations through the different trophic levels, is often not available.
- There is a greater agreement on the protection objective (the ecological function) and on an approach based on the protection of a percentage of species in a theoretical ecosystem. However, the

methodology applied depends on the availability of data, which in turn determines whether point estimate or probabilistic approaches are used.

The review also indicates:

- A lack of toxicity data and standardised and ecologically meaningful analytical methods for toxicity assessment.
- A lack of data on bioavailability for the different species.
- The need to develop ecotoxicology theory in order to be able to define what is safe for ecosystems or for specific ecological functions, taking into account their resilience and resistance.
- The need to incorporate appropriate ecological functions into the derivation of human health S/G values. These depend on landuse and mainly relate to phytotoxicity.

Derivation of values aimed at protecting the quality of water resources

Two main steps are typically included. The first is an assessment of water quality, usually based on established guidelines and criteria (e.g. WHO or country guidelines) for drinking water, or other specific water uses. The second is an evaluation of migration pathways. Both leaching and run-off models may be considered in the assessment; expert judgement is used to decide on dilution factors between source and receiving water.

Policy issues

In Volume 2 of this work,[5] the main policy differences between countries are reviewed with regard to soil and groundwater protection and remediation, as well as other key issues affecting the definition of S/G values. It is interesting to note that some of the policy differences depend on factors such as legal and administrative systems, land ownership arrangements, industrial histories, social and cultural aspects (such as the perceived seriousness of soil pollution), land and water use including pressure for its re-use, and economic aspects such as the availability of resources and technology.

Also included in Volume 2 is a summary of the legal and administrative documents dealing with soil protection, or with the remediation of contaminated sites in participating countries. Detailed analysis of these documents is beyond the scope of this project, but a summary is included because it may be helpful in understanding the philosophy and policy objectives behind the definition of S/G values. In some coun-

tries specific legal frameworks have been drawn up in which S/G values are used. In the absence of such a framework other laws are used, especially those concerning waste (a major cause of soil pollution) and water (one of the natural resources most directly affected by soil contamination).

With respect to risk assessment and remedial action approaches, historically it has been possible to distinguish those countries advocating the concept of *multifunctionality* from those favouring the idea of *fitness for use* (see Chapter 2). The Netherlands and Denmark have traditionally defended multifunctionality, but from a practical point of view the fitness for use approach was often applied in these countries as well as in others. Ultimately both philosophies appear to reflect objectives shared by all participating countries: multifunctionality as a long-term goal and fitness for use as a pragmatic strategy for sustainable regeneration of brownfield sites. Some of the factors influencing this kind of decision are the current availability of cost-effective decontamination technologies, the need to reduce development pressures on greenfield sites, and the desirability of reducing the burden of aftercare.

With regard to the environmental receptors considered, there are no major differences between countries. The most important relates to the implicit protection of water resources (mainly groundwater) and to the level of protection of ecosystems. In most countries screening/guideline values are specifically derived for protection of human health. Screening/guideline values for ecosystem protection, which apply to specific landuses where ecological attributes are the key issue for concern (natural use, sensitive ecological areas, etc.), have been derived in five countries. In Sweden and Canada some degree of protection of ecosystems is considered necessary for adequate soil functioning, and its fitness for any given landuse, and is therefore included in the derivation. Equally, S/G values for the protection of water resources have been derived, for example in Sweden, Flanders (for agricultural land use), Canada, the Netherlands and Denmark.

The list of priority contaminants considered is broadly similar in most countries and includes eight main groups of compounds: metals, other inorganic compounds, organic volatile compounds, polycyclic aromatic hydrocarbons, aliphatic hydrocarbons, chlorinated solvents, other chlorinated compounds and pesticides. With regard to landuse, five generic scenarios are usually considered (residential with gardens, residential without gardens, parks and recreational areas, agriculture and commercial/industrial use); however definitions vary somewhat from country to country, depending on culture and lifestyle. Tables are

included in Volume 2 comparing exposure routes and other key parameters used in the derivation of S/G values in the various countries. Moreover, variations in geology, climate and soil type between countries affect the S/G values derived.

It may be concluded that many factors affecting the definition and derivation of S/G values will differ among the various European countries. Complete harmonisation of soil and groundwater policies would be difficult and, indeed, undesirable, given the huge variations in physical and cultural factors that have a bearing on risk assessment of contaminated sites. The main objective of this chapter has been to facilitate comparisons between countries and to help avoid misinterpretations or misuse of the different criteria and values. Nevertheless, there is general concurrence on the desirability of agreeing on the technical and scientific issues behind the derivation of S/G values, and on the value of international collaboration in research.

8.4 Concluding remarks

The topic of screening/guideline values is not specifically scientific, and most of the scientific issues that underpin the derivation of such values have been covered in other chapters. Nevertheless, the topic straddles the science–policy interface and raises some important practical issues:

- An agreed scientific basis for the derivation of S/G values is still evolving, especially for ecologically-based values. The paucity of basic data, and absence of a well-developed theory, are major stumbling blocks limiting the more widespread application of ecological risk assessment. Moreover, the need to consider ecological function even in those landuses where humans are the main receptors emphasises the need to develop, and translate into practical terms, concepts such as full-ecological function and partial ecological function.
- The need to deal with different sources of uncertainty and variability. It is worth pointing out that many of the issues raised in the context of S/G values (e.g. the reliability of uncertainty factors and the effectiveness and robustness of modelling procedures) are just as relevant for traditional site-specific risk assessment.
- The need to minimise the gap between actual exposure conditions and generic assumptions, especially in relation to worst case *versus* realistic approaches, point estimates *versus* probabilistic functions for different parameters, and restrictions on the applicability of values. Moreover, it seems necessary to develop appropriate statistical tools

to assist in the use and interpretation of S/G values. These tools should be considered in the context of:

- the derivation process: handling of variability and uncertainty;
- establishment of site contaminant concentrations and their spatial variation, which will be used for comparison with screening/ guideline values;
- the social and economic implications of the values, and the need to avoid them being thought of as *magic numbers*;
- cost–benefit analysis of the different approaches;
- consistency of values for different media as part of integrated environmental objectives;
- the derivation of values for mixtures of chemicals, grouped according to their likely joint occurrence on contaminated sites.

References

1. Siegrist, R.L. (1900) Development and implementation of soil quality and clean-up criteria for contaminated sites. In *Contaminated Soil* '90 (F. Arendt *et al.*, eds). Kluwer, Dordrecht.
2. EPA (1996) *Soil Screening Guidance: Technical Background Document*, EPA/540/R-95/128. Office of Solid Waste and Emergency Response, Washington, DC.
3. ICRCL (Interdepartmental Committee on the Redevelopment of Contaminated Land) (1979). *Acceptable Levels of Toxic Elements in Soils,* ICRCL 16/78. Department of the Environment, London.
4. VROM (1983). *Soil Protection Guideline.* 1983/1990. Staatsuitgeverij, The Hague.
5. Ferguson, C. *et al.* (in preparation) *Risk Assessment for Contaminated Sites in Europe*, vol. 2, *Policy Frameworks.* LQM Press, Nottingham.

Chapter 9

Better methods for risk assessment

9.1 Scientific and research needs

Contaminated land risk assessment is still underpinned largely by scientific research done for other purposes. The nature of the assessment is to a large extent determined by the availability of these more or less useable scientific building blocks. Whether current assessment procedures really address the question of *risk* in a rigorous, quantitative way may be questioned. Further development and integration of the building blocks needed for risk assessment is of the utmost importance if assessment is to be more than a mere sequencing of separate disciplines like soil and water sampling, chemical analysis, exposure modelling and toxicology. In a fully integrated approach, choices of toxicological endpoints must have consequences for the design of sampling schemes and exposure models, and vice versa. Uncertainties at each stage in the assessment should be recognised and may lead to the use of probabilistic or other techniques for dealing with uncertainty. Decision-support systems may provide guidance for risk managers to help balance reduction of uncertainties against the costs of additional investigation. Integrated risk assessment procedures have yet to be fully developed, and progress will depend on research in two main areas:

- The nature of contaminated land, which deals with the identification and analysis of pollution and its impact on human health, water resources and other environmental receptors; and
- The relationship between soil and water contamination and *fitness for use,* which specifies the conditions for sustainable landuse in urban and rural areas.

The nature of contaminated land

This research area includes the development of techniques, methods and procedures to assess soil and water pollution (and their relationship) and to establish the scale and intensity of the pollution in such a way that the consequences for landuse and environmental protection can be assessed. Soil and groundwater pollution cannot be described by a set of fixed parameters. Pollutants may degrade, disperse and transform with time. Risks might decrease or increase in time, depending on landuse, soil and aquifer characteristics. The dynamic interplay between these factors must be understood in order to predict future impacts, to keep polluted areas under control, and to assess various options for remediation. Three interlinked themes for research may be distinguished: site characterisation, protection of water resources and bioavailability.

Site characterisation

Site investigations should provide much of the data necessary for exposure analysis and risk assessment, and must also quantify the uncertainties associated with site characterisation. The linking of site investigation to exposure analysis and evaluation of uncertainties needs further development in most countries, and requires scientific research in the following areas.

Robust and rapid low-cost techniques for investigating potentially contaminated sites. Robust and low-cost techniques must be based on necessary and sufficient data. These data need to be collected during investigations for human toxicology, ecological risk assessment, fate and transport analysis and modelling. Better techniques need to be identified, developed and field-tested. Often a phased approach is used in site investigation, so assessment tools must be developed at various levels of sophistication. For preliminary investigation more extensive use of non-intrusive investigation methods is highly desirable. The use of geostatistics and regionalised variable theory should be explored further.

Improved methods for estimating and interpreting the accuracy and likely variability of the whole sampling and analytical process. Two aspects are important here: first, the quantification of accuracy and variability, and second their control by quality assurance. Quality assurance and quality control are important for all aspects of site characterisation to ensure reliable results. The need to assess variability and accuracy may encourage the use of statistical approaches that do justice to the notion of risk as a probabilistic concept. The value of prob-

abilistic risk modelling may critically depend on the statistical soundness of the results of site investigations.

Methods that yield information at spatial scales relevant for exposure assessment. At present contaminated land is investigated with methods primarily derived from soil mapping. Although generally considered a useful starting point, these methods do not yield all the information needed for assessing exposure to soil pollution. Depending on the heterogeneity of the soil and the spatial scale at which various exposure routes operate, information may be needed on a more refined scale than current methods seem to provide. Information on the spatial scales most relevant to human and ecological risk is generally lacking, and is of the utmost importance for the development of integrated risk assessment procedures.

Characterisation by biosensors and bioassays. Current practice in contaminated land risk assessment starts with a chemical characterisation of the site. Biological effects and risks are then assessed by interpreting the chemical data in biological or toxicological terms. The use of bioassays and biosensors may provide a shortcut in this procedure, and may be very cost effective. Apart from the mere development of biological test methods or indicators, a frame of reference for the interpretation of bioassay and biosensor results is also lacking. Comparative studies, where biological methods are combined with chemical assessment, might provide such a frame of reference and encourage the use of cost-effective biological methods.

Measurement and modelling of gas-phase contaminants in soil and buildings. An important exposure mechanism, particularly in connection with the redevelopment of industrial land for housing, is the transport of airborne contaminants (vapour or dust) from soil to indoor air. This topic has been the subject of recent research in the Netherlands, the UK, Australia and the USA but is still very under-researched relative to many other areas. Specific research needs include the validation of transport models for benzene and particle-bound metals (e.g. lead), the treatment of spatial and temporal variations in these processes, and the control of exposure to airborne contaminants during remediation works.

Protection of water resources

In most countries groundwater is protected as a resource that should remain pure, as implied by the EU Groundwater Directive. There may

be situations, however, where the application of this principle in environmental groundwater protection has become impossible due to the extent and persistence of contamination. In these situations a more risk-orientated approach may be used. Methods to predict whether soil pollution will in the long run migrate to groundwater, and to what extent groundwater pollution will disperse and affect abstracted or surface water quality, are of the utmost importance. Current practice is mostly based on geohydrological models. A broader scientific basis including geological, geotechnical and probabilistic approaches may yield substantial improvements. In particular, the transport of contaminants in the unsaturated upper layer of the soil and the behaviour of contaminants at the interface between the unsaturated and saturated zones both need further study. The following issues may be addressed:

Macropore transport and fate of contaminants in soil. Transport of contaminants in soil and groundwater is often calculated based on the assumption of bulk flow through a homogeneous porous medium. However, in many cases the preferential flow pattern is through fractures or other macropores in soil, the most severe implication being much faster transport rates of contaminants. Therefore, there is a need to study the mechanisms governing the fate and transport of contaminants in such macropores, including the importance of colloids, partitioning processes in macropores, bacterial colonisation behaviour, etc. Groundwater flow in fractured aquifers is already the subject of intensive study.

Organic carbon as a major factor governing fate and transport. Organic carbon in soil is known to play a key role in both transport and fate of organic contaminants in soil and groundwater, e.g. as a medium for sorption, or as a co-substrate for microbial processes. Until recently organic carbon has been looked upon as having the same characteristics in all cases. However, organic carbon may differ substantially in chemical composition and properties, which may have a strong influence on its ability to serve as a co-factor in the fate and transport processes. Methods to separate, characterise and differentiate between different types and fractions of organic carbon in soil are urgently needed, as is an understanding of the dynamics of organic carbon in contaminated soils.

Methods to assess the natural potential of soil and rock to attenuate contaminants, and techniques to monitor the process. Knowledge of biodegradation of contaminants is substantially based on research performed under controlled conditions in the laboratory. Experience

with contaminated land suggests that many biodegradable substances may be much more persistent in the real world. On the other hand, some substances that appear to be persistent in the laboratory have turned out to degrade slowly in nature. The potential for soils and rocks to biodegrade certain substances depends on the specific characteristics of the site. It is not a characteristic of the pollutant alone. Because natural attenuation is both pollutant dependent and site dependent it is difficult to predict. Yet such predictions are of great importance in risk assessment, especially with regard to the way risks change with time, and for remediation strategies. The practical feasibility of using extensive and low-cost biological remediation depends on the availability of reliable assessment methods.

Modelling interactive metabolism of contaminants. Reactive modelling in most cases addresses biodegradation only at one point as a function of a number of geochemical, geological and hydrogeological parameters. However, in many cases the different contaminants are degraded sequentially as a result of redox gradients down the flow path or due to variable degradability of the contaminants under the prevailing environmental conditions, including co-metabolic dependencies etc. To better reflect field situations models must be able to handle a higher degree of complexity than is currently possible.

The interaction and general fate of contaminant mixtures. In current risk assessment practice each polluting substance is usually considered in isolation. In toxicology it is well appreciated that one toxic substance may increase or reduce the effect of other substances. Similarly, contaminants might also interact in the environment, influencing each other's fate and transport. Interaction between contaminants, especially for transport and fate of complex mixtures like petroleum hydrocarbons, is an important subject for further research.

Free phase fate and transport. Most research has dealt with fate and transport of contaminants in solute form. Classical methods for the prediction of contaminant migration fail when pollutants do not mix with groundwater but exist as floating or sinking layers of free phase liquid, the so called LNAPLs and DNAPLs. Special methods must be developed to help characterise these products in the subsurface and to predict future behaviour. A better understanding of how free-phase products affect soil processes must also be achieved. One topic is the solubilisation kinetics of free-phase liquids as a possible factor governing their fate and transport. Another topic is the biodegradation of free-phase

contamination, including the microbial toxicity of free phase and the colonisation of microbial populations on free-phase surfaces.

Bioavailability of contaminants in soil and groundwater

Bioavailability is a facet of the interaction between organisms (soil fauna, bacteria, plants) and their chemical environment. Soil characteristics partly determine bioavailability for organisms, and organisms in turn create their own environment by influencing soil properties. Current bioavailability research is too focused on abiotic aspects. Organisms are often modelled as a special form of soil organic matter which is exposed to water in the pore spaces of soil, and which does not respond to changes in the environment. Future research should critically test the applicability of simple abiotic bioavailability modelling and should consider the biology of the organisms involved more explicitly.

Another aspect that is not fully appreciated is that bioavailability may change with time. More research on ageing processes of polluted soils and on time dependence of bioavailability should be encouraged. Progress in this field should lead to cost-effective procedures for determining bioavailability of compounds as they exist in the environment.

Fitness for use

Human health risks

The primary need among contaminated land risk assessors is for human toxicity data that adequately reflect the chemical forms, modes of delivery, exposure conditions and bioavailability found in the context of contaminated sites. It is recognised, however, that the quality and relevance of fundamental epidemiological and toxicological data are severely constrained by both cost and ethical considerations. Realistic research needs are therefore summarised below.

To develop the theoretical basis and practical tools (decision-support systems) necessary to allow relative risk contributions to be taken into account when setting target values for soil contaminants. There is a need to test and demonstrate with specific examples how a methodology and decision-support system can be used to develop soil target values that take into account the relative contributions of soil and non-soil as well as site and non-site sources to total exposure. In the UK this approach has already been used in the derivation of the draft soil guideline values for lead. These values have been derived such that any

required reduction in soil-lead values is proportional to the contribution that soil makes to total lead uptake in two-year-old children living in urban environments.

To identify areas where significant improvements in epidemiological and toxicological understanding can be achieved at a realistic cost. There is a particular need to understand better how contaminant–matrix interactions affect the bioavailability of contaminants after entering the human body, how to predict the availability of pollutants within the human body, and the difference between intake and uptake. In order to arrive at a realistic description of human toxicity the availability of contaminants in soil relative to the availability in the animal experiments used to derive reference values (e.g. TDI) must be taken into account. Research in this area is badly needed.

To develop better and more consistent ways of interpreting currently available toxicological and epidemiological data, and dealing with the associated uncertainties. Dealing with combined exposure to mixtures, and finding a more scientifically defensible basis for uncertainty factors are prominent issues. These are needs common to all toxicologists, not just those working in the context of contaminated land. However, it would be beneficial to develop a more consistent decision-support methodology among participating countries (and, indeed, worldwide). This would help stakeholders distinguish better between scientific factors and social, economic and political factors underlying decisions.

To increase the predictive power of exposure models and specify human risks in space and time. Validation of models for various exposure pathways is important in view of their impact on contaminated land decision making. Although risk assessment procedures cannot usually be tested empirically as a whole, testing of specific parts is still possible. Further studies of exposure pathways from soil to humans will also yield information on the appropriate spatial scale for human exposure. This is important for the design of soil sampling schemes. For better risk assessments it will also be necessary to consider the appropriate time frame for risks in view of:

- choice of exposure period and averaging period
- degradation of pollutants.

Risk comparison

Many of the above themes relate to the issue of risk comparison. Research in this area may be seen as a key step in addressing the basic

question: how significant are the risks associated with contaminated sites in relation to other risks, and on what factors do these judgements of significance depend? This area of research requires an innovative integration of scientific risk assessment methodologies and those of the social and behavioural sciences.

In addition, the *valuation* of risks and risk management options is a multidisciplinary field involving many areas of risk study, including remediation economics, insurance, law, ethics and policy. An important task here is to complement traditional cost–benefit and risk–benefit analyses with modern multicriteria decision methods.

Ecological risk assessment

Whereas human health risks concern the health of an individual, ecological risk has to address the health of populations of a multitude of species and ecosystems. Ecological risk is still based on the No Observed Effect Concentration (NOEC) concept and results of toxicity testing in the laboratory. There is at present no ecosystem theory that can serve as a framework for interpretation of NOEC data. Although human health risk assessment is also largely based on laboratory experiments with animals, there is a framework for interpretation in medicine, sociology and psychology, which is lacking in the ecological approach.

Many forms of landuse by humans also need a certain level of ecological functioning in soils, sometimes referred to as the life support system. In the derivation of landuse-based remediation goals, discussions about human toxicity dominate and the requirements of the life support system are often neglected. If more ecological research were devoted to the life support system concept this problem could be adequately addressed.

Another neglected ecological field is groundwater ecology. Groundwater reserves are under pressure from over-exploitation, and in some countries water shortages are already occurring. At present groundwater is protected as a source of drinking water. The ecological consequences of groundwater pollution are still poorly understood and would provide additional motives for groundwater protection.

The main topics for research in the field of ecological risk assessment are:

- impact of a site on the surrounding environment
- ecological recovery at the site
- changes in community structure caused by pollution-induced tolerance *versus* classical ecotoxicological endpoints
- biomagnification risk and adverse effect on food chains

- ecological soil and groundwater quality requirements related to human landuse.

Models for risk assessment

Models are powerful tools for integrating the various elements in a risk assessment, e.g. site characterisation, fate and transport of contaminants, exposure assessment and risk estimation. They may be used as tools for site-specific assessment of a given contaminated site, or to derive generic screening or guideline values. Models, however, are abstract representations of complex systems, and are based on numerous assumptions. It is therefore of the utmost importance that models and submodels should be validated and tested in real-world situations, either in contaminated land risk assessments or in special research projects. Field-testing and validation of models raises important questions about the precision and accuracy of model predictions. In particular, can we expect accurate estimates from the overall assessment in view of the many uncertainties in source characterisation, in exposure assessment and in the toxicological basis for tolerable daily intakes?

From a general methodological point of view an important area for research might be a study of how risks estimated from site-specific exposure modelling differ from those estimated using generic criteria. What do the results of an assessment actually mean? And how does exceeding a toxicological reference intake or soil screening level relate to the probability of human health or ecological effects occurring? From a risk characterisation point of view it is important to know how accurate one could hope to be on the probability of an effect occurring, as well as on the magnitude of the effect. This in turn would influence risk communication.

Risk perception and communication

Use of the results of scientific risk assessment in environmental decision making must take the perception of various risks and other social issues into account. The development of coherent risk communication strategies is important: How should we communicate the results of risk assessment and the choice of a solution to those who are or feel themselves to be at risk as a consequence of (potentially) contaminated land? And how should we communicate with other stakeholders whose perceptions may be very different?

The following questions might be addressed:

- How should one decide on the relative importance of a negative effect which is unlikely but has very significant consequences, as opposed to one which is more likely but has minor consequences? Is this always a site-specific judgement or could a general framework for deciding on relative importance be established?
- Can we develop a general measurement scale for the importance of adverse effects and for *acceptable* or *tolerable* probabilities of occurrence in view of the different perception between the general public and those who might suffer the adverse effect?
- Are risks at contaminated sites caused by human activity more or less acceptable than naturally occurring risks?

9.2 Other needs

A large amount of research dealing with the scientific building blocks of risk assessment has been reviewed in the previous chapters. Risk assessment, risk analysis, policy making and decision making are also extensively studied in the social sciences and in psychology. Attempts[1,2] to integrate the scientific and technical framework and socio-psychological aspects of risk analysis have had limited success, but a decision theoretic approach might yield valuable results.

Risk assessment for contaminated sites is a rather loose assemblage of concepts and methods borrowed from various disciplines. Until recently, the research community seems to have had little interest in studying fundamental issues related to integrating the various building blocks of contaminated land risk assessment. Developments in this area are being driven by regulators who need better decision-support systems. The limitations of toxicological reference values, exposure modelling and soil and groundwater sampling are not widely understood, especially by the generalist type of scientist or engineer often involved in site investigation and risk assessment.

Risk assessment is not yet recognised as a coherent scientific discipline. Further integration of the building blocks will be achieved under pressure from environmental policy makers with the support of industry. International cooperation is important in this for a number of reasons:

- to avoid unnecessary duplication;
- to provide a wider basis for scientific peer review;
- to provide a common database for physico-chemical and other basic data;

- to promote international cooperation on the assessment of toxicity of substances in soil and groundwater;
- to promote mutual understanding of the way science is put to work in developing and delivering national policies.

International cooperation in environmental science and policy is at present considered necessary to solve large-scale problems. Some people feel that soil and groundwater problems are local problems and therefore international cooperation is not so important. This is a rather naive point of view. Global problems evidently need political solutions at an international level. Local problems need solutions that reflect local needs and circumstances, but this does not mean that international exchange of ideas about how to tackle these problems is of limited value. Reinventing solutions for soil and groundwater problems in each country is simply a waste of time and money. Common political solutions may not be necessary or desirable, but exchange of technical and scientific approaches between countries is extremely valuable.

Improving risk assessment for contaminated sites depends not only on the results of research projects. Other requirements have to be met, of which the most important are:

- training of risk assessors and decision makers;
- networks for communicating new approaches and practical experiences;
- linking fundamental science to real-world problems.

Training of risk assessors and decision makers

The science behind risk assessment has to be applied in a local decision-making context. This means that local decision makers need to have sufficient grasp of the scientific basis and/or to manage the input from specialist consultants. In most countries there is a substantial need for training local contaminated land risk managers and others involved in decision making. This is especially so in the light of the many variabilities, uncertainties and other methodological pitfalls in current risk assessment approaches.

Risk assessment only leads to defensible decisions if its limitations are recognised. Sometimes the risk assessment cannot provide all the answers, and *risky decisions* may have to be taken. This is not new in environmental policy, at least at the national level. However, the decisions relating to soil and groundwater pollution are often taken by local authorities with limited expertise to draw on. Training of local authority personnel is therefore very important.

Network for communication of new approaches and practical experiences

Even the best-trained personnel have to maintain and update their knowledge. They will also gain practical experience with various forms of risk assessment in different situations. These experiences may be very beneficial to others. The perspectives for dealing effectively with contaminated land are broader if they are based on shared knowledge and do not depend critically on the experience of an individual assessor. Most countries also need a platform for sharing information, developing case studies, disseminating new approaches and identifying research priorities. In view of the many scientific questions in risk assessment, and the fact that present day science is performed in an international context, an international network for the scientific aspects of contaminated land risk assessment brings great potential benefits.

Linking fundamental science to real-world problems

Fundamental science traditionally aims at establishing general theories that are tested against observations under controlled and defined circumstances. Very often, theories are difficult to apply directly in complex areas like soils and ecosystems. A long period of applied research is needed before most products of fundamental science can be used effectively in real-world problems such as the assessment of risks from contaminated land.

A more promising strategy is not to wait until contaminated land problems catch the attention of the fundamental scientist, and the outputs of fundamental scientists catch the attention of the applied scientist, but to find ways of bringing all types of scientist together in the context of contaminated soil and groundwater risk assessment. The solutions generated may be less fundamental and universal than the products of fundamental sciences, but they may be more directly applicable. Generalities and new scientific theory may evolve as practical experience with contaminated land accumulates. One of the best ways to improve risk assessment is to link fundamental and applied RTD projects with specific contaminated land problems.

References

1. Vlek, C.A.J. (1990) Beslissen over risicoacceptatie. *Gezondheidsraad* A 90/10, Den Haag.
2. Royal Society (1992) *Risk: Analysis, Perception and Management*. The Royal Society, London (ISBN 0-85403-467-6).

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