CHAPTER 1 PRINCIPLES AND CONCEPTS IN ECOLOGICAL RISK ASSESSMENT

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1.1 Scope and content

Numerous decisions have to be made before, during and after conducting an ecological risk assessment. This book is an attempt to guide risk assessors and stakeholders in the decision making process. Its main objectives are to:

1. Present and describe the basic structure of a decision support system (DSS) for assessing site specific risk from contaminated land to ecosystems.
2. List, review and provide further references for useful tools for site specific risk assessment with special attention to bioavailability.
3. Present a basic flow chart for assisting in the selection of risk assessment tools.
4. Present and recommend ways to weight, scale and use the results of the tests in a DSS.

This book represents the final outcome of the EU project Liberation (Development of a decision support system for sustainable management of contaminated land by linking bioavailability, ecological risk and ground water pollution of organic pollutants, EVK1-CT-2001-00105). Although special attention was given to organic pollutants, most of the conclusions, the choice of bioassays, scaling of results etc. can also be used in the case of heavy metal contaminated sites provided adjusted tools for estimating bioavailability of heavy metals are applied. Useful and more detailed information about heavy metals and possibilities to assess their bioavailability can be found in e.g. Peijnenburg et al. (1997), Plette et al. (1999), Zhang et al. (2001), Ehlers and Luthy (2003) and Van Straalen et al. (2005).

This DSS is not a comprehensive “all-you-need-to-have” document for managing risk of contaminated land, as it focuses primarily on supporting decisions made when assessing risk to the terrestrial environment. It addresses only indirectly the risk to ground water and associated (connected) fresh water systems. Nevertheless information about e.g. reduced bioavailability may be useful when assessing potential risk for leaching of contaminants to ground water or fresh water. Recognising these limitations is important in the decision making process. Therefore, risk assessors focusing on the ecological impact of a contaminated site need to collaborate closely with risk assessors concerned about the risk to humans and ground water. Moreover, all risk assessors need to have close contact with the relevant stakeholders and other parties involved in the management and the current and future use of the site.

This book is organised as follows: after a brief introduction to the overall principles and concepts in ecological risk assessment and decision support systems in Chapter
One, Chapter Two presents the challenges and solutions for including bioavailability of organic pollutants in an ecological risk assessment framework. In Chapter Three there is a more detailed presentation of decision support systems for evaluating environmental risk of contaminated soil including a description of the different stages of the Ecological Risk Assessment (ERA) process. Chapter Four gives a more detailed description of the principles in the Triad approach, which we recommend as a powerful tool in the final stage of the ERA. Chapter Five provides guidance for a tiered assessment of ecological risk of contaminated soil including decision charts and selecting of site-specific assays in the various tiers. Chapter Six contains a presentation of the most common and useful tools for assessing fate and effect of pollutants found at contaminated sites. The tools are organised in various toolboxes each allocated to a specific tier of the ERA process. Finally, Chapter Seven is a short review of the outcome of using the Triad approach in a case-study with a contaminated site in Denmark.

1.2 Ecological risk assessment in brief

Ecological risk assessment (ERA) is a process of collecting, organising and analysing environmental data to estimate the risk of contamination for ecosystems. Assessing the ecological risk of contaminated soil is, however, a complicated task with many obstacles associated with the process. Terrestrial ecological risk assessment was developed later than aquatic risk assessment, and is furthermore complicated by the fact that soil systems are very heterogeneous. As a consequence of this soil pollution is often less uniform and more difficult to define than fresh water ecosystems. Moreover, land and hence soil is typically private property and traded as real estate. Professional and economically divergence between the interests of land-owners, scientists, national authorities, engineers, managers, lawyers, NGOs and regulators is therefore not unusual.

There are typically two major types of ERA. The first one is predictive and often associated with the authorisation and handling of hazardous substances like pesticides or new and existing chemicals in the European Union. This type of ERA is ideally undertaken prior to environmental release of the substance. The second type of ERA is a description or estimation of changes in populations or ecosystems at specific sites or areas already polluted and should hence be conveyed as impact assessment rather than risk assessment. The principles of ecological risk assessment are described in numerous review papers and books, e.g. Ferguson et al. (1998), US-EPA (1998), Suter et al. (2000), Lanno (2003), Weeks et al. (2004) and Thompson et al. (2005).

Often ERA is performed in phases or tiers, which may include predictive as well as descriptive methods. The successive tiers require, as a rule of thumb, more time, effort and money. The paradigm for ERA may vary considerable but is typically based on an initial problem formulation based on a preliminary site characterisation, a screening assessment and a risk characterisation followed by risk management.
Prior to actually performing an ecological risk assessment it has to be decided whether (cheaper) alternatives are available. The size and the location of the contaminated area has to be evaluated in order to assess whether, for example, a simple dig-and-dump exercise would solve the problem at a much lower cost. This may be the case of very small plots within a larger area. On the other hand, remediation could be out of the question due to logistical problems, e.g. physical deterioration of a larger area in order to clean up a smaller area or disturbance of protected species.

In most European countries the first stage of the ERA of contaminated soils consists of rather simplified approaches including comparison of soil concentrations with soil screening levels (SSL, also known as quality objectives, quality criteria, benchmarks, guideline values etc.). Methodologies for deriving SSLs are described elsewhere, e.g. Wagner and Løkke (1991) and Posthuma et al. (2002). One of the keystones in deriving environmental quality criteria is the use of standardised test procedures. A collection of terrestrial soil tests can be found in for example Sheppard et al. (1992), Keddy et al. (1994), Van Gestel and Van Straalen (1994), Tarradellas et al. (1997), Løkke and Van Gestel (1998). Furthermore, a list of guidelines can be found at www.iso.org and www.oecd.org.

Soil screening levels (SSL) are common and very useful screening tools for assessing ecological risk. However, large discrepancies have been observed between effect levels derived from spiked experiments conducted in the laboratory, i.e. SSL, and the effect levels found when organisms are exposed to soil collected at contaminated sites or when monitored in the field. Many reasons are given for the contradiction between observed toxicity, mobility and degradation rates in freshly spiked soil and the observation made in the field. One explanation frequently given is a reduction in bioavailability over time as pollutants become sequestered into the soil matrix. This phenomenon is particularly important for a sound evaluation of risk and remediation options for contaminated sites. Chapter Two gives a short introduction to the problems and solutions associated with ageing of soil contamination.

Although single species laboratory tests with spiked materials have their obvious benefits, e.g. they measure direct toxicity of chemicals and interpretation is therefore simple, other supplementary tools are often needed to assess risk. Bioassays, performed with contaminated soil ex situ, are one of the more frequently used higher tier alternatives. Bioassays have the advantage, compared to the use of spiked soil samples, that the toxicity of a specific soil may be assessed directly. Bioassay testing has therefore the ability to account inherently for the complete mixture of contaminants, including degradation products and metabolites, in the sample. This is important, as generic calculations of the combined effect of mixed contamination are prone to large uncertainties. Furthermore, the in situ bioavailability of that specific soil is (at least almost) maintained in the laboratory during the exposure period. Consequently, bioassays are often considered more realistic tools for risk assessment than generic soil screening levels based on spiked laboratory soils.
A number of uncertainties or limitations may, however, be associated with the use of bioassays and the interpretation of the results obtained in these. The test species are exposed to the polluted soil for a short period as compared to the permanent exposure condition found at contaminated sites. Furthermore, beside the presence of contaminants, test species are exposed in an (almost) optimal environment as stress factors like predators, inter- and intra-species competition, drought, frost and food depletion are eliminated during the controlled exposure in the laboratory. Finally, a limited number of species are available for testing.

Contaminated soil may be assessed using multi-species tests, lysometers or terrestrial model ecosystems (TME). In these, species interaction may be evaluated by introducing several species to the systems or monitoring the endogenous populations. Natural climatic conditions may be included if the test systems is kept out-doors.

A crucial issue when analysing the result of bioassays, TME and field studies in all tiers of ERA is the presence or absence of a proper reference site or soil. This is true for chemical information (i.e. background levels in that region), toxicological data from bioassays (i.e. site relevant reference soil and control soil in order to verify the test performance) and ecological field surveys. The reference soil should in principle resemble the contaminated soil in all relevant parameters, e.g. texture, pH, organic matter, water-holding capacity, nutrient content. Since matched reference soils are often lacking this is a practical problem that often is difficult to solve and hence should be considered and discussed in detail before initiating the ERA. The lack of reference sites in field surveys may, however, in some cases be solved by the use of multivariate techniques (e.g. Kedwards et al., 1999ab), which relate the species composition and abundance to gradients of pollutants. Increased computer power and the presence of new easy-to-use soft-ware tools have increased the possibility to move away from more conventional uni-variate statistics like ANOVA to more powerful multivariate statistics, which use all collected data to evaluate effects at a higher level of organisation. Statistical methods like the power analysis may also be very useful in planning and designing large-scale ecotoxicity studies like mesocosms, TME or field surveys (Kennedy et al., 1999).

### 1.3 Concepts of ecological risk assessment

Schemes, paradigms or programs for conducting ecological risk assessment of contaminated sites have been developed in a number of countries or regions. It is beyond the scope of this book to present all of these in detail. The outline of ERA paradigms is presented below for a few selected countries, i.e. USA, the Netherlands, United Kingdom and Canada.

The US Superfund program for assessing ecological risk is one of the first and also most developed initiatives on ERA of contaminated sites. The Superfund program aims at quantifying potential effects on human health and ecological risks at inac-
tive hazardous waste sites. The Superfund risk assessments determine how threatening a hazardous waste site is to human health and the environment. Risk assessors should seek to determine a safe level for each potentially dangerous contaminant present. For humans, this is a level at which adverse health effects are unlikely and the probability of cancer is very small. For ecological receptors, determining the level of risk is more complicated, as it is a function of the receptors of concern, the nature of the adverse effects caused by the contaminants, and the desired condition of the ecological resources.

The US-EPA has published an Ecological Risk Assessment Guidance, which should be followed when assessing risks at Superfund sites. As all sites are considered unique this should always be done in a site-specific manner.

The ERA process suggested by the US-EPA for Superfund sites follows an eight step process, which can be broken down into four categories, i.e. 1) planning and scoping, 2) problem formulation, 3) stressor response and exposure analysis and 4) risk characterisation. An overview of the eight steps is presented in Figure 1.1 (more details can be found on the web pages of the US-EPA). Essential for all steps are a negotiation and agreement of the need for further action between the risk assessor, the risk manager and other stakeholders, the so-called scientific-management decision points (SMDP).

SMDP made at the end of the screening-level assessment will not set an initial clean-up goal. Instead, hazard quotients, derived in this step, are used to help determine potential risk. Thus, requiring a cleanup based solely on those values would not be very likely, although it is technically feasible. There are three possible decisions at the SMDP:

1. There is enough information to conclude that ecological risks are very low or non-existent, and therefore there is no need to clean up the site on the basis of ecological risk.
2. The information is not adequate to make a decision at this point, and the ecological risk assessment process will proceed.
3. The information indicates a potential for adverse ecological effects, and a more thorough study is necessary.

In the Netherlands contaminated sites are first determined using a set of soil screening levels called target and intervention values, which take both human and ecological risks into account (Swartjes, 1999; VROM, 2000; Rutgers and Den Besten, 2005). At seriously contaminated sites remediation or other soil management decisions are required if the risks cannot be neglected based on a site-specific ecological and human risk assessment, and the chance for dispersion of the contaminants. Until now, the ecological risk assessment has been based on chemical analysis, including a Decision Table harbouring critical dimensions of the impacted area (Table 3.3). Recently, an up-date of the soil protection act is foreseen (VROM, 2003), clearing the way for a site-specific risk assessment based on the Triad (Rutgers and Den Besten, 2005).
The United Kingdom and Canada have also developed framework for ecological risk assessment of contamination land (e.g. CCME, 1997a; Weeks et al., 2004). A cornerstone in the UK framework of ERA is the connection to the statutory regime for identification and control of land potentially affected by contamination, also known as Part IIA of the Environmental Protection Act of 1990. This act defines land as contaminated if:

- A contaminant source and a pathway along which the contaminant can move is present and the contaminant (potentially) can affect a specified receptor.
- There is a significant possibility of significant harm.
- Pollution of controlled waters is occurring or is likely to occur.

![Figure 1.1 The eight steps in the US-EPA framework for risk assessment of contaminated Superfund sites. DQO = data quality objectives](image-url)
Currently only risk to controlled waters and certain protected habitats (defined in Part IIA) are covered. The Framework does, however, also address how to perform ERA in areas not currently covered by Part IIA. The UK framework is based on schemes found in e.g. USA, Canada and the Netherlands. Like these it is a based on a tiered approach where the initial Tier 0 aims to determine whether a site falls under the Part IIA of the legislation. It involves the development of a Conceptual Site Model (CSM), which described what is already (historically) known about the site, e.g. whether there is a likely source-pathway-receptor linkage.

The conceptual site model is followed by an initial screening phase (Tier 1) and an actual site-specific characterisation (Tier 2). Tier 1 is a simple deterministic comparison of chemical residue data and the soil quality guideline values supplemented with simple soil-specific toxicity testing. The final step (Tier 3) involves more detailed in-situ studies and for example ecological modelling based on a more advanced ecological theory. Tier 3 is not likely to be conducted at many sites. More details on for example the selection of tests in the various Tiers can be found in Weeks et al. (2004) and on the web page of the Environment agency in UK (www.environment-agency.gov.uk).

General and technical information about the framework for ERA of contaminated land in Canada can be obtained from the homepage of the Canadian Council of Ministers of the Environment (CCME) (www.ccme.ca) or in two reports from CCME published in 1996 and 1997 (CCME, 1996; 1997a). In the Canadian approach soil quality guidelines are derived for four different land-uses, i.e. agricultural, residential/parkland, commercial and industrial. More information about the paradigms for deriving the Canadian soil quality guidelines can be obtained in CCME (1997b).

1.4 Principles and concepts for decision support systems

Chapter 3 to 5 in this book describes a newly developed system to support the decision-making process when assessing the risk of contaminated land to the environment. The target of the risk assessment is to provide the assessor with an objective and scientific evaluation of the likelihood of unacceptable impacts of the contaminants to the environment. It is, however, recognised that guidance on how to assess environmental risk is only part of the whole picture when stakeholders face the challenge of handling and managing a potentially contaminated site.

Decisions on how to manage the risk at a particular site depends on a number of parallel considerations. The outcome of the risk assessment (may or may not) form the foundation for taking a decision regarding appropriate risk management options. Identifying suitable technical solutions to soil contamination for example is not solely based on the outcome of the risk assessment. Risk management options needs therefore to be addressed in a holistic manner. Key factors include aspects such as the driving forces for remediation, e.g. environmental or human risk, recrea-
tional value or urban development plans, and availability, feasibility, suitability and cost of technical solutions to remediation.

Generally the decision-making process for contaminated land management includes several phases (Bardos et al., 2003), e.g.

- An identification phase in which the (magnitude of) problem is identified.
- A development phase in which likely solutions are identified and (further) developed.
- A selection phase in which the solution is chosen and implemented.
- A monitoring phase in which the effectiveness of the above decisions are examined and evaluated.

The DSS on environmental risk assessment in this book is clearly meant to support the identification phase as outlined above. However, the outcome of the process, e.g. which organisms are at risk and where and when they are at risk, may be very useful when, at a later stage, suitable options for remediation have to be identified and selected.

For a more comprehensive discussion about DSS the reader is referred to a review performed by CLARINET (Contaminated Land Rehabilitation Network for Environmental Technologies) under the 5th Framework Programme of EU and published by the Austrian Federal Environment Agency in 2003 (Bardos et al., 2003).