Chapter 15
Site-Specific Ecological Risk Assessment

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Abstract In many countries, soil quality is expressed in chemical concentrations as Soil Quality Standards to address the potential ecological risks in a first tier of an Ecological Risk Assessment (ERA). In cases where application of these standards do not provide satisfactory results, additional tools are required. In this chapter the focus is on these tools, i.e. ERA taking into account the complete mixture of contaminants and the integration of data from bioassays and field ecological observations according to a weight of evidence approach. A straightforward Triad framework, combining three lines of evidence, was introduced in the Netherlands in 2007 and is presented here.

Contents

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15.1 The Soil Ecosystem and Site-Specific Risk Assessment

Site-specific Ecological Risk Assessment (ERA) is a process aiming at the support of site management decisions with respect to contamination (Suter et al. 2000). Typically, site-specific ERA focuses on a specific site. A broad spectrum of decisions can be considered, such as adaptive land management, changes in land use, and tailoring the site remediation objectives. In order to arrive at these decision points, data have to be collected, organized and analyzed to estimate the risk of contamination for ecosystems. ERA encompasses a complex procedure as many issues have to be addressed. A comprehensive ERA requires various contributions from stakeholders, authorities, managers and experts at different stages of the process before it can be fully accomplished.

Site-specific ERA ranges from rather simple or small situations to very complicated processes with many experts involved and numerous data evaluations conducted, often leading to tailored decisions. The commonality of different ERAs arises from a persuasive notion of adverse ecological effects, irrespective of the complexity or dimensions of the site. Any ERA should start with the application of generic and conservative principles for optimum protection (first tier Risk Assessment). This may be accomplished, for instance, by comparing contaminant concentrations at the site with national Soil Quality Standards, which may be adjusted for differences in soil characteristics and background concentrations (a common practice in the Netherlands, see Chapter 1 by Swartjes, this book). For the majority of sites such a generic Risk Assessment is sufficient to exclude unacceptable risks. However for a number of sites the uncertainty in this kind of generic and general assessment may be too high, e.g. when the Soil Quality Standards do not provide the right insight or the Soil Quality Standards are exceeded. This will often trigger more site-specific and less-generic actions, in higher tier Risk Assessment. In this stage, divergence between experts may occur, because different investigations/disciplines may not provide similar conclusions. Divergence between authorities and stakeholders may also reveal as a result of soil – or rather land – being treated as real estate with fixed boundaries, while contamination and ecological damage typically cross such site boundaries. Therefore, ERA should be embedded in structured frameworks allowing complex paradigms to be developed and the outcomes to be transparent, uniform and applicable for contaminated site management decisions (Barnthouse 2008; Hope 2006; Linkov et al. 2006).

For ERA in terrestrial systems, lessons can be learned from aquatic and sediment systems (Chapman and Anderson 2005; Rutgers and Den Besten 2005). Terrestrial systems, however, differ because they are generally more heterogeneous, have much slower dynamics, their food web characteristics are yet unidentified, and the contamination characterization may be less strictly assessed through the complex impact of the soil matrix on ecological effects. As a result, uncertainty is a key issue that needs specific emphasis in terrestrial ERA.

For these reasons it is essential that ERA is organized in phases, or tiers, including predictive as well as descriptive methods in order to reduce these uncertainties in a practical way. The successive tiers require increased inputs and, as a rule of thumb, more time, effort and money. The paradigm for ERA in specific cases may vary considerably, but typically includes an initial problem formulation based on a preliminary site characterization, followed by a tiered risk characterization, and it ends with a list of Risk Management options.

The question of what to consider as an aspect of the ecosystem needing consideration in an ERA is not so complex as one might think. Indeed, Egler (1977) has stated: 'ecosystems are not more complex than you think, they are more complex than you can think'. This notion automatically provides a rationale for simplification, i.e. it is justified to address only a few aspects which should be documented, rather than deliberately trying to 'catch it all'. Consequently, it is better to report on A risk, instead of THE risk of contamination (Rutgers et al. 2000). Secondly, aspects needing consideration may vary from very broad and general to site-specific peculiarities. In the Netherlands any ERA starts with a broad and conservative assessment via application of Environmental Quality Criteria aimed at protection of the complete ecosystem. This relates to the protection of both biodiversity and ecological functions, which is obtained through the application of so-called Species Sensitivity Distributions (SSDs; Posthuma et al. 2002; see Chapter 14 by Posthuma and Suter, this book) for species and processes (Sijm et al. 2002). In addition, rather conservative thresholds are applied for protection targets and remediation targets, i.e. 95 and 50% respectively (Swartjes 1999). Thirdly, the ecosystem approach may be broken down in distinct steps, by addressing different aspects at different levels of integration. For instance, in the case of the terrestrial environment the focus might be spread over four important aspects connected to carbon and energy transmission (Fairbrother et al. 2002):

1. primary production, i.e. focusing on organisms performing photosynthesis, e.g. green algae and plants;
2. fragmentation, i.e. focus on organisms involved in the cutting and grinding of large organic fragments and organic macromolecules e.g. earthworms and micro-arthropods;
3. decomposition and mineralization, i.e. a focus on the final breakdown and synthesis of organic components in the soil e.g. micro-organisms, protozoa and worms (earthworms and pot worms);
4. consumption and predation. i.e. a focus on (the stability of) interactions between organisms in so-called trophic webs. e.g. nematodes and micro-arthropods.

So, it is defendable to restrict ERA in the earlier tiers to, for instance, these four generalized aspects. In the latter tiers it is defendable to extend the ERA by including specific species such as protected wild life (nature) and ornamental plants (parks, gardens).

15.1.1 Appreciation of the Ecosystem at Contaminated Sites

Before any site-specific investigation is initiated, it is important, as the *first step* in an ERA, to evaluate whether there are any ecological concerns associated to this specific site (Fig. 15.1, in which the *Framework for site-specific ERA* is given).

In most countries, no detailed and systematic inventory has been made of how often ecological concerns could be associated to contaminated sites. This is for example true for Denmark. Denmark has for decades collected data and generated a comprehensive and relatively complete record of the contaminated sites within the country (Danish EPA 2008). To date this inventory has registered approximately 24,000 contaminated or potentially contaminated sites. However, a screening of how frequently valuable ecosystems, e.g. *Natura 2000 areas*, are located on contaminated sites was not initiated until recently. The investigated area covered one of the five Danish Regions. Here a total of more than 600 contaminated sites were located at – or in very close vicinity of – an important conservation nature area (terrestrial, fresh water and marine) corresponding to approximately 10% of the contaminated sites in the region. The dominating sources of contamination in these areas were tar from coating of fishing nets in the late history, shooting ranges and dump sites.

A comprehensive study in the Netherlands has shown that out of 500,000 suspicious locations, approximately 28,400 potentially contaminated sites are located within recognized nature areas or *Natura 2000 areas* (Versluijs et al. 2007). It is expected that 3,200 sites in these areas have to be remediated, comprising a total surface area of about 8,400 ha.

In the Netherlands the approach to ERA might be different from many other countries with a soil protection policy. A Risk Assessment for the terrestrial ecosystem applies for all sites with a serious soil contamination, and remediation should be seriously considered for all unpaved and uncovered soil, including those at industrial sites. In this sense, the ecosystem has the same status in the Risk Assessment as human health and the chance of dispersion and spreading of the contaminants (Swartjes 1999; Versluijs et al. 2007). This policy pays tribute to the notion that soil harbors important natural functions, which are essential for mankind. Consequently, human and ecological risk may trigger remediation at contaminated sites for all land uses, albeit the thresholds differ.

15.1.2 Stakeholder Involvement

The *second and the third step* in the site-specific ERA would then be to select relevant stakeholders and experts for the steering committee and the team of risk assessors (Fig. 15.1). The size and shape of such a Steering committee and risk assessor team depends on the type and magnitude of work anticipated for the

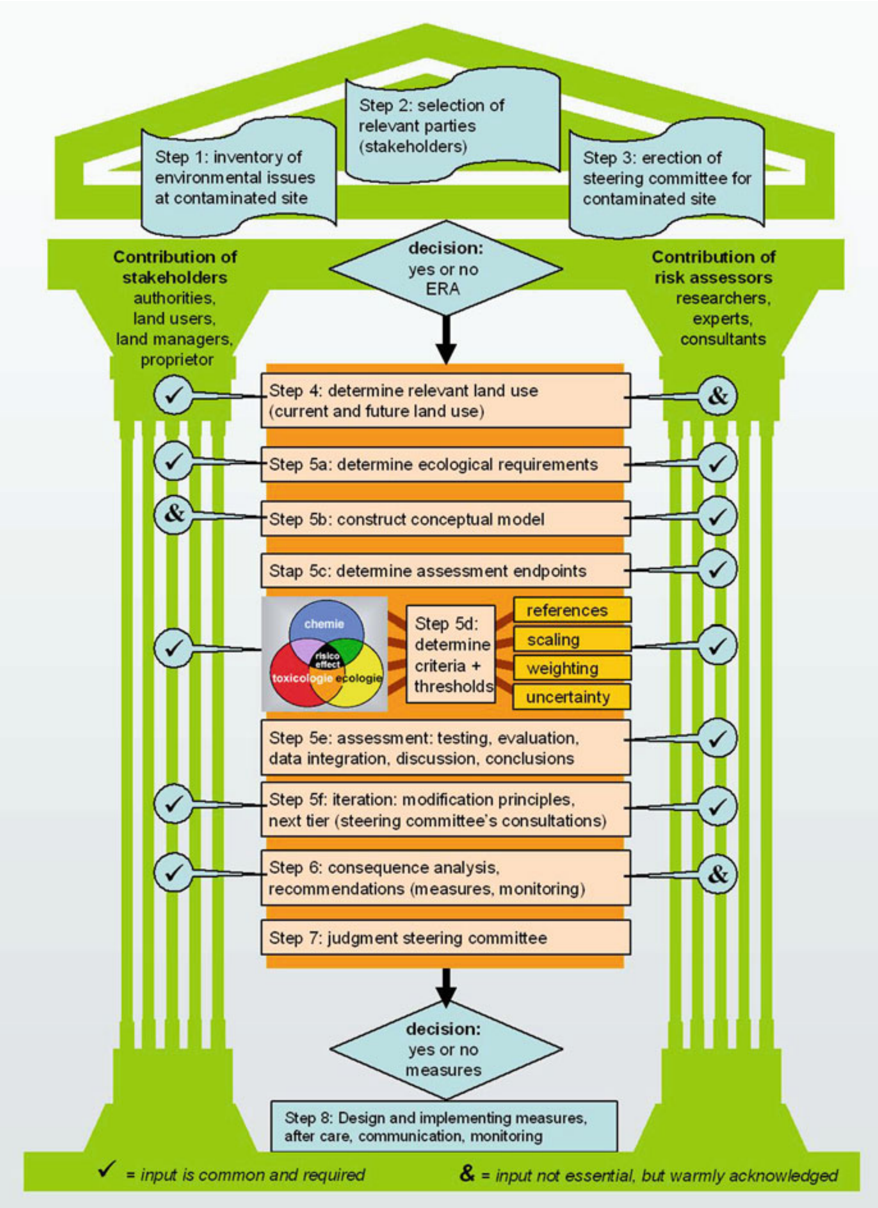

Fig. 15.1 Framework for site-specific ERA depicted in stylized portal. The pillars represent contributions from stakeholders and land users (*left*) and risk assessors and experts (*right*). After the decision to start up ERA, a steering committee should perform guidance and evaluations. The steps 4 and 5 can be used as guidance on the assessment pathway. The subsequent step 6 is included in order to facilitate the incorporation of remediation options. Step 7 comprises an independent judgment from a peer review. The framework was based on an earlier publication (Rutgers et al. 2000), slightly modified and is currently incorporated in a procedural standard of the Netherlands Normalization Institute (NEN 2010)

respective site. However, it is important to involve a wide range of stakeholders early in the process, in order to reach a mutual understanding and acceptance of the conceptual site model for the terrestrial ecosystem, including the target of protection and means of successful risk mitigation prior to initiating any actual investigations. Stakeholder involvement should therefore include contaminated-site experts from authorities, land users and land managers/owners. The team of risk assessors should include people from academia and consultancies, capable of performing ERA, covering various field of expertise.

The Steering committee should then, in alliance with the risk assessors, determine the land use as the *4th step* in the ERA. Subsequently, the actual site-specific Risk Assessment is initiated as the *5th step* of the ERA by identifying the ecological requirements related to the specific land use (5a). In the subsequent steps (5b, 5c and 5d), a listing of relevant assessment endpoints for each of the identified ecological requirements needs to be identified and agreed upon, i.e. if nutrient cycles and plant biodiversity are considered important in relation to this specific land use, a suite of tools, bioassays or monitoring end-points should address the related endpoints. Examples of relevant assessment endpoints, like sensitive crops, key species, decomposition and nutrient cycles for various land uses can be found in Jensen and Mesman (2006).

15.2 Working Hypotheses, Definition of Conceptual Models and ERA Frameworks

Contributions from and interactions by risk assessors and risk managers are essential in the definition of the conceptual model and working hypotheses. In the conceptual model a simplification of the real system is obtained in order to frame the results of the Risk Assessment. The conceptual model contains two key elements (US-EPA 1998): (i) a set of working hypotheses and (ii) a diagram representing the links between the working hypotheses. Consequently, the conceptual model sets the limits of the ERA. Terrestrial ecosystems are complex and dynamic systems. It therefore requires a well elaborated conceptual model to reduce complexity and integrate system attributes in order to develop clear solutions and management decisions. A unifying ecosystem theory is lacking, for example, making the selection of assessment end-points difficult. ERA can focus on specific endpoints, like the protection of particular species (e.g. endangered species, wild life) or the performance of Ecosystem Services of the soil system (e.g. nutrient cycling, Natural Attenuation, water retention, etc). However, ERA can in principle also cover risks derived from a more ethical perspective of environmental protection. Consequently, all biotic elements will be potentially useful to some extent.

Working hypotheses are specific assumptions about potential risk to assessment endpoints (US-EPA 1998). They are formulated on the basis of one or more information sources like contamination history and data, professional judgments and information from the ecosystem at risk. Working hypotheses are important elements in ERA because they improve the level of site specificity compared to the generic application of Soil Quality Standards. Usually, adverse effects of contaminants on ecosystem attributes are formulated in terms of the pathway, from the presence of contaminants to the potential adverse ecosystem effects, i.e. the source – pathway – receptor links. This conceptual model operates on the working hypothesis of an established pathway between source and receptor.

Conceptual models may range from very simple to rather sophisticated and complex models. A very simple and commonly used conceptual model relates to the derivation of Soil Quality Standards, where the corresponding working hypothesis is that all organisms are equally important in the ecosystem (Posthuma et al. 2002). More sophisticated conceptual models have also been used (e.g. Baird et al. 2008; Bennett et al. 2007; Faber 2006), for instance with modifications in Exposure Assessment (e.g. including bioavailability considerations) or in the end-points (e.g. field birds) and ecological processes. Regardless of the level of detail, these models will always embody a simplification of the actual ecosystem. Once conceptual models are formulated, they serve as a framework for the selection of tools and for the definition of thresholds in the assessment.

Any set of tools for ERA should be embedded in a decision-making framework, which primarily consists of phases such as initial problem and scoping phases, exposure and Hazard Assessment and Risk Characterization. Several decision-making frameworks have emerged in the literature, basically following the same outline. The US-EPA has published one of the more advanced frameworks (US-EPA 1998), including many later amendments (Barnthouse 2008; Suter et al. 2000). Also in Canada and Europe, frameworks were published (CCME 1996; Faber 2006; Jensen and Mesman 2006; Weeks and Comber 2005). In the Netherlands such a framework is recently accepted in a procedural standard (Fig. 15.1).

In this chapter we will not review and discuss various frameworks for ERA, but instead focus on a few practical issues related to the application of additional tools in a *weight of evidence* (WoE) approach.

When all these important first steps have been fully discussed, the actual Risk Assessment procedure can start. The next paragraphs will describe one of the most

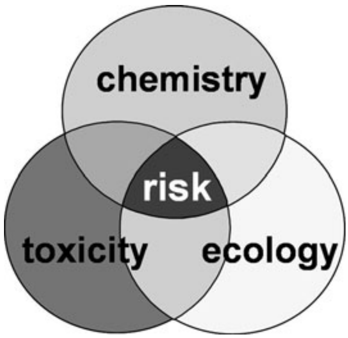

Fig. 15.2 Schematic presentation of a weight of evidence approach using the Triad. The three independent lines of evidence consist of a chemical-based assessment, a toxicity based assessment using bioassays, and an ecological assessment using data from ecological field surveys

operational and reliable methodologies, i.e. the *Triad approach* which combines three *lines of evidence* (Fig. 15.2).

15.3 Weight of Evidence and the Triad Approach

In order to deal with uncertainties in the process of ERA in a pragmatic and responsible way, it has been proposed to use a weight of evidence (WoE) approach (Chapman et al. 2002; Hull and Swanson 2006; Long and Chapman 1985; Rutgers and Den Besten 2005; Suter et al. 2000). The rationale is that multiple and independent ways to arrive at the same type of conclusions will provide a stronger 'evidence' for ecological effects, substantially improving the reliability of ERA. Unfortunately, precise definitions and application schemes of WoE in ERA are unclear (Weed 2005), but a series of papers edited by Chapman et al. (2002) addressed several issues. In this chapter we will not focus on clarifying these issues, but instead we present relevant scientific developments and practical considerations for the application of a WoE at contaminated sites (Critto et al. 2007; Dagnino et al. 2008; Faber 2006; Jensen and Mesman 2006; Weeks and Comber 2005). In addition, we illustrate these considerations by a newly adopted framework in the Netherlands (Mesman et al. 2007; Rutgers et al. 2008b).

For terrestrial ecosystems, WoE approaches are still in an exploration and developing stage (Critto et al. 2007; Jensen and Mesman 2006; Rutgers and Den Besten 2005; Semenzin et al. 2007, 2008; Suter et al. 2000). The Triad approach relates to a specific form of a WoE (Fig. 15.1; step 5d). It is based on the simultaneous deployment of three independent types of assessment tools:

- site-specific *chemical characterization* often combined with the estimation of ecotoxicological effects using literature data, e.g. by calculating a risk index;
- application of bioassays or biomarkers in order to determine de novo and ex situ *toxicity* in soil samples from the site;
- on-site *ecological observations* or other monitoring data that provide insight in the plausible effects of the contamination.

The major assumption is that WoE using a combination of tools from these three independent disciplines will lead to a more detailed and correct assessment than an approach, which is solely based on one of these, for example the total concentrations of contaminants at the site. A multidisciplinary approach will thereby help to minimize the chance on false positive (incorrectly assuming that there are effects, whilst in reality there are no effects) and false negative (incorrectly assuming that there are no effects) conclusions.

The advantage of the Triad approach can also be stated as follows: the combination of three simple instruments enables the reduction of model uncertainties, which is compatible or better than reducing model uncertainties using one sophisticated tool. Information about model uncertainties can be deduced from results of tools from different disciplines, rather than from one set of tools in one discipline. This makes the Triad approach a scientifically sound and practical instrument, during different stages of ERA.

15.4 Practical Issues for Adoption of the Triad Approach

15.4.1 Uncertainty

Uncertainty is a key element in Risk Assessment and should be properly addressed and communicated. Uncertainty can be seen as the state of imperfection within the total available amount of information with respect to the environmental problem and the requested decision to be made in time (Walker et al. 2003). Uncertainty contains both subjective and objective elements. The subjectivity originates from the judgment about the validity and appropriateness of the information. Objectivity comes from the data and facts related to the contaminated site. Uncertainty is therefore often separated in:

- Variability and error, i.e. lacking or imperfect data and data from systemic variations in space and time. An example is the variation in results by application of a specific tool: in one assay with real replicates, with pseudo replicates, from inter-laboratory variation, and through gradients in space or time.
- Incertitude, i.e. model imperfections; or in more popular terms: you do not know what you do not know. This uncertainty is demonstrated by application of different tools at one occasion (sample or site), both within a line of evidence, or between different lines of evidence.

It is important to realize that both types of uncertainty need appropriate, but inherently different approaches in the Risk Assessment. Recently, linguistic uncertainty was introduced additionally to these two types of uncertainty (Carey and Burgman 2008; Levin 2006). Linguistic or language related uncertainty between risk assessors and risk managers may arise especially in the case of ERA, because of a lack of appropriate terms and definitions, imprecise problem framing and different perspectives and views on the environment (Kellett et al. 2007). A clear and transparent communication between stakeholders before, during and after the execution of an ERA is therefore crucial: it is the only way to minimize the chance on misperceptions.

It is beyond the scope of this chapter to recapitulate all aspects of uncertainty. Instead references are made to the respective literature, (e.g. Beer 2006; Burton et al. 2002a, 2002b; Levin 2006; Nayak and Kundu 2001; Walker et al. 2003). As an idea we state that variability and error are primarily solved by increasing the amount of effort, e.g. via more samples, more replicates, and further optimizing the noise to signal ratio via improving of assessment tools. Weight of evidence approaches like the Triad seems to be preferred in order to reduce uncertainty caused by incertitude (including ignorance and indeterminacy).

15.4.2 Selection of Assessment Tools

The application of the Triad approach comprises the selection of tools which:

(1) fit in the specific tier of interest (from screening-level to highly sophisticated tools);
(2) cover all three lines of evidence;
(3) effectively address the selected end-points.

The final suite of selected tools should allow for dealing with and ultimately reducing uncertainty in ERA. A tool is defined as an instrument for quantification of a specified aspect of the ecosystem. The outcome must ultimately be expressed in a one dimensional number indicating the level of ecological effect on the uniform scale. Tools range from very simple (a screening level bioassays or a concentration plus literature toxicity data) to highly sophisticated and integrated (results from BLM modeling, maturity index of nematodes or a food web stability index). Any tool must be based on site-specific information through modeling and/or measurements and on information from literature data and ecotoxicological reasoning for interpretation of the data on the uniform effect scale (see below for an explanation in Section 15.4.3).

Elaborating on the three issues for selecting tools (see former paragraph):

- Sub 1 (tiers). With respect to the tailoring of the specific tier of interest, standardization and costs of analysis are important issues for selection of tools, especially in the lower tiers. Yet, even screening tools should be sufficiently reliable and sensitive to demonstrate effects of contaminants under field realistic conditions. Finally the tools should be relevant for the ecosystem under investigation. More sophisticated and elaborated tools are used for improving site specificity in the higher tiers of the ERA.
- Sub 2 (lines of evidence). Each tier of the Triad approach should cover three independent types of assessment tools, representing three different lines of evidence. This requires at least one tool for a chemical based assessment (*chemical characterization*), at least one bioassay (*determination of toxicity*), and at least one type of on-site ecosystem observation, which can be related to effects of contamination (*ecological observations*). When the different lines of evidence are comparable in terms of effort and matching level of insight, a balanced weighting between the lines of evidence can be applied (see below for more details on weighting of the results).
- Sub 3 (addressing selected end-points). The appropriateness of respective tools to serve as indicators for selected endpoints is the third and last issue. The tools should provide insight about compliance of end-points with respect to the potential effects of the contamination at the site. Many ecologically relevant end-points cannot be directly assessed, because of imperfect knowledge and lack of tools. Instead models or surrogate systems are used to extrapolate from the assessment tools to real world situations. Confirmation of ecological significance of the individual test systems originates from track records or literature evidence of the

respective tools in comparative cases. If not, this should be specifically addressed. This is often the case with tools used in higher tier Risk Assessment, because of insufficient scientific foundation.

As a rule of thumb, simple, common, standardized and low-cost tests should be used in the lower tiers of the ERA, whereas more laborious and sophisticated tests should be applied in the higher tiers. Guidance for selecting appropriate tools is available (e.g. Fairbrother et al. 2002; Jensen and Mesman 2006; Römbke et al. 2006a; Rutgers et al. 2008a). The highest level of protocol standardization of tests is reached in international guideline programs such as ISO and OECD. Whereas the OECD test program has focused on tests suitable for the evaluation of chemicals, the ISO guideline program has, at least recently, focused on test systems for the evaluation of the risks of contaminated soil (Römbke et al. 2006b). Additional protocol standardization comes from quality assurance systems like Good Laboratory Practice (GLP). Methods described in the scientific literature can also be used, especially in the higher tiers of ERA. In any case, it may frequently be necessary to adopt the tests to site-specific conditions. The number of laboratories able to perform a specific test on a routine or semi-routine basis is another issue when selecting the set of tools. Finally, the acceptance of the tests by the stakeholders, risk assessors and the scientific community plays a role in the selection criteria. Last but not least, it is important to minimize the bias caused by the risk assessor's background. It is, for example, a human commonality to overstress the importance of the own expertise in solving complex problems. With large multidisciplinary research teams, however, this problem is somewhat reduced.

The combination of different tools from different disciplines at different levels of standardization, robustness, sensitivity and ecological relevance without being able to fit them all in one comprehensive and accepted ecosystem theory is in fact a matter of combining 'apples and oranges'. This highlights the need for a proper *multi-criteria decision analysis* (MCDA) in ERA (Chapman et al. 2002; Linkov et al. 2006). With MCDA it is possible to combine different pieces of not a-priory related information in an unconstrained way. It opens the possibility to cope with divergent, but *best professional judgments* from separate experts in a transparent framework process.

15.4.3 Quantification and Scaling

Essentially, the results from all tools to be applied, including bioassays and ecological field surveys, should be funneled into the ERA. Key for efficient use of information is 'scaling'; i.e. the projection of results from different tools on a common and unified 'effect scale' (e.g. inhibition of growth, or loss of reproduction should be both expressed in the comparable units as an ecological effect). The primary aim is to maximize the utilization of individual results, and to use results from all tests together in transparent and integrative schemes, for example in a decision matrix. Burton et al. (2002a, b) reviewed several possibilities for disseminating final WoE findings, and concluded that tabular decision matrices are the most transparent and quantitative representations.

Ideally, ERA for aquatic, sediment and terrestrial systems should follow the same set of conventions for scaling. In practice, however, there are slight differences for the following reasons:

1. There is a wide range of standardization levels and terrestrial methods differ in sensitivity, making it difficult to define one set of homogeneous 'rules' for interpretation. Although initial thoughts for scaling of e.g. bioassays, biomarkers and community-level end-points are obtained from *best professional judgments*, still much experience is lacking. It is expected that these rules can be obtained step by step from the building up of practical experience from ERAs at contaminated sites.
2. Interpretation of test results in terms of 'effect' or 'no-effect' inevitably will result in the loss of valuable quantitative information. Except for the situation for ERA in surface water and sediment systems, the limited experience with use of the Triad approach for terrestrial systems demands for exploration and efficient use of virtual all available information in a quantitative manner.
3. In aquatic systems toxicity can be determined after a pre-concentration step, allowing the application of relatively insensitive tools and producing fewer false negative results. It is virtually impossible to concentrate soil samples putting higher demands on tools and the use of results in ERA.

For evaluation and integration of the results from the three lines of evidence in the Triad (chemistry, toxicity, ecology) a *quantitative decision matrix* is constructed. To this purpose, it is necessary to use a uniform *effect scale* for the quantification of each of the separate effect levels in the Triad approach, running from zero (no effect) up to 1 (maximal ecosystem effect). Consequently, the results from each tool (bioassay, biomarker or ecological field survey) should be projected on this effect scale, according to best available knowledge from the literature or best professional judgments (BPJ) from consulted experts. Useful and advanced examples of scaling rules and the construction of such a *quantitative decision matrix* can be found in Jensen and Mesman (2006), Dagnino et al. (2008) and Semenzin et al. (2008).

Different tools will obviously require different approaches. For instance, for a growth test the percentage of inhibition can be implicitly used as the measure for effects. For ecological field monitoring, however, the results should be scaled relatively to the ecological state of a reference site (= 0), and a (theoretical) state indicating 100% effects. Information from field monitoring is often composed from multiple variables putting specific demands on the scaling of multi-dimensional information to a one-dimensional effect value (Jensen and Mesman 2006).

Furthermore, the method of scaling should account for limitations in working range of an assessment tool with respect to the effect scale. This is sometimes denoted as the biological scale of the measurements (e.g. Gaudet et al. 1995; Wright and Welbourn 2002). The effect scale is usually defined on the level of populations of protected species, whole communities, ecosystem functions or some kind of Ecosystem Integrity (Suter et al. 2000). However, assessment tools addressing subcellular responses like biomarkers are rather sensitive and can be perfectly used as early warning signals, but have a limited range on the effect scale, i.e. a relatively low signal. On the contrary, field surveys at the population or community level are less sensitive but generally 'closer' to the assessment endpoints, making the response on the effect scale much stronger (closer to 1).

In cases with large and wide-spanning datasets, it might be feasible to apply a suite of indices in order to take advantage of all data in the best suitable way (Dagnino et al. 2008). Examples of such indices are:

- Environmental Risk Index (ERI): Quantifying the level of biological damage at the population level, comparable to the Triad with similar legs.
- Biological Vulnerability Index (BVI): Using e.g. biomarkers to assess the potential ecosystem stress and threats to biological equilibria.
- Genotoxicity Index (GTI): Used to screen for genotoxicity effects.

Whereas the first index is assigned to the ecological leg of the Triad, the two latter are assigned to the ecotoxicological leg.

These indexes were used on a site-specific case in north Italy by Dagnino et al. (2008). They showed that the Triad-based decision system (Environmental Risk Index) as well as the biomarker-based index, identified the two contaminated industrial areas as under risk. However, in contrast to the result from the Environmental Risk Index, the results from the biomarker studies, i.e. the BVI, indicated that also the chosen low-contamination site was under stress and that in some of the sampling occasions, the GTI index at this site was comparable to the contaminated industrial sites, indicating a general stress syndrome in soil organisms from that region.

These different indices allow for an elaboration on the plausible links between causes and effects. Finally, when one answer is required to aid contaminated site assessment and management decisions, these three indices should indicate adverse ecological damage on the uniform effect scale (0–1 effect scale) too.

Projection of test results on the uniform effect scale requires a certain level of experience. This expertise is *fundamental* to ERA, the importance of it can not be overlooked. Without sufficient expertise one cannot expect a responsible underpinning of the decisions from the site-specific Risk Assessment. When the issue of scaling is properly and responsibly solved, the information from separate tools from individual disciplines can be effectively used together in ERA. Fortunately, the WoE approach will help to address mismatches of specific scaling methods due to wrong assumptions (Chapman et al. 2002). Together with ecotoxicological reasoning, this information can than be used to correct the scaling method of respective tools. Accordingly, lower tiers in the Triad approach should contain tests which are, to some extent, standardized, while at higher tiers the comparative less-standardized tests should play a role in order to improve the level of site-specificity.

Once all results are quantified in the uniform effect scale, the overall response of a set of (biological) methods can be calculated. To this purpose, a weighting algorithm of different test results is required. This is described in Section 15.4.4.

15.4.4 Weighting of Effect Values

Besides the issue of scaling, the risk assessor should pay attention to the issue of weighting. Weighting applies to different tools, i.e. weighting within a line of evidence, and applies to weighting across different lines of evidence in the Triad approach. Some general principles apply to this. As a default, the three lines of evidence in the Triad should be equally weighted. Each line of evidence has its own weaknesses and strengths. However, together they form the strongest basis for ERA according to the principles of a balanced WoE approach. In specific cases, specific considerations demand for a differential weighting between the different lines of evidence in the Triad approach. The absence of adequate reference sites is typically the most problematic with ecological field surveys at strongly disturbed sites. In these cases, ecosystem changes might dominantly be caused by other factors than soil contamination. Another example of differential weighting is a difficult chemical assessment, because of complicated exposure routes and limited toxicity data. In that case it is defendable to give a lesser weight to the chemical based assessment (*chemical characterization*) than to the two effect values from the other lines of evidence (*determination of toxicity* and *ecological observations*).

Within one line of evidence attention should be given to a suite of aspects within the ecosystem. Typically, the starting point is an equal weight for all organisms and processes, applying the following popularized statement: 'All organisms are unequal, but equally important'. Another possibility is to collate data in different trophic groups like primary producers, decomposers of organic matter (fragmentation and mineralization) and consumers, and give these different trophic groups equal weights. Within any individual line of evidence of the Triad approach, differential weighting of results may be applied for three possible reasons:

1. Ecological considerations, e.g. from different land use classes, may trigger a differential weighting, which should be defined in the conceptual model. This allows extra attention to specific (functional) groups, key species, endangered species, 'charismatic' species or even specific ecological processes in the terrestrial ecosystem.
2. Differential weights can be applied in order to account for the uncertainty or variation within the end-points. Tests with a high level of uncertainty, or with a high variation in results, may be given a smaller weight in the ERA (Menzie et al. 1996).
3. Differential weights might be applied in order to correct for biases in the expected number of false positive or false negative results. For instance, the geometric mean of the inverted effect value gives extra weight to those observations with a positive response. This acknowledges the fact that many bioassays or ecological field surveys are sometimes not able to demonstrate ecological effects on the screening level, although in reality these effects are present (false negatives). This is especially a problem with tight budgets or highly dynamic systems, because the number of replicates is often too limited for demonstrating significant effects.

Den Besten et al. (1995) used differential weights in the ERA for aquatic systems following a multi-criteria decision analysis. Effects on e.g. top predators and benthos received a higher weight than parameters such as mouthpart (mentum) deformities. This information was used to rank different sites according to their possible ecological risks. For the terrestrial system, less experience is available. Semenzin et al. (2007, 2008) and Critto et al. (2007) developed tabular decision matrices to address the issue of weighting.

15.4.5 Reference Information

A crucial issue when analyzing the results of bioassays or ecological field observations in different tiers of ERA is the *reference information*. This information can be gathered from reference sites, reference samples, or literature data. Of course, analysis of reference sites and reference samples is preferred, since this optimizes the site-specificity in ERA. Due to a lack of sites and samples, literature data may partially substitute a lack of suitable references. Rutgers et al. (2008a) recently published reference data of soil system attributes for ten common combinations of land use and soil type in the Netherlands, which may be used as a source of reference information for ERA.

The issue of reference data is relevant for any line of evidence in the Triad approach, i.e. chemical characterization (i.e. background levels in that region), toxicological data from bioassays (i.e. reference soil for quantification of the no-effect level and control soil in order to verify the test performance) and ecological field surveys (i.e. the ecological status of reference sites). The perfect reference response resembles the response from the contaminated soil in all relevant aspects, besides the effects from soil contamination. When a site contains gradients in soil characteristics, also multiple references have to be gathered in order to reflect this gradient. To reach this goal, parameters that may affect test performance, like the soil's texture, pH, organic matter, humidity and available nutrients, should be verified between contaminated and reference soils. Sometimes, information is available about the influence of soil characteristics on test performance (e.g. Natal-da-Luz et al. 2008). It often is a practical problem to identify matching soil samples. This problem has to be tackled in a sensible way and hence should be considered and discussed in detail before initiating the ERA. The lack of suitable reference sites in field surveys may, however, statistically be solved by the use of multivariate techniques (e.g. Kedwards et al. 1999), which relate the species composition and abundance to gradients of contaminant concentrations in soil, taking into account possible effects of other factors ('confounders'). However, such an approach needs the analysis of large numbers of samples in order to account for all possible gradients that may shape the ecological parameters in the survey (Rutgers 2008). Many software tools are available and have increased the possibility to use powerful multivariate analysis, which use all collected data to evaluate effects at a higher level of organization. Of course, in a strict sense, causal inference of field effects from contaminants is impossible, due to imperfect reference information (Boivin et al. 2006; Everitt and Dunn 2001; Jensen and Pedersen 2006; Rutgers 2008).

15.5 Integration of Lines of Evidence and Final Results

After the results have been scaled for each test, it is possible to integrate the results of the different tests in each line of evidence. In Table 15.1 an example of collecting and presenting the data from a Triad-based ERA is given. In order to integrate

Table 15.1 Example of a table for collecting, summarizing and integrating data from a Triad-based ERA

Triad aspect	Parameter	Weight factor	Sample A	Sample B	Sample C
Chemistry	Sum TP total concentrations	1	0.00	0.76	0.92
	Sum TP porewater concentrations	1	0.00	0.62	0.75
	effect (chemistry)		**0.00**	**0.70**	**0.86**
Toxicology	Microtox	1	0.36	0.21	0.70
	Earthworm test	1	0.00	0.00	0.52
	Germination test	1	0.00	0.05	0.20
	effect (toxicity)		**0.14**	**0.09**	**0.30**
Ecology	Nematode community analysis	1	0.00	0.50	0.55
	Microbial parameters	1	0.00	0.25	0.45
	Micro-arthropod community analysis	1	0.00	0.15	0.32
	Plant community analysis	1	0.00	0.00	0.60
	Earthworms	1	0.00	0.45	0.24
	effect (ecology)		**0.00**	**0.29**	**0.45**
	Effect assessment chemistry	1	**0.00**	**0.70**	**0.86**
	Effect assessment toxicology	1	**0.14**	**0.09**	**0.51**
	Effect assessment ecology	1	**0.00**	**0.29**	**0.45**
	Integrated assessment (risk)		**0.05**	**0.42**	**0.67**
	deviation		0.14	0.55	0.38

In a first step the data are grouped per line of evidence, i.e. chemistry, bioassays and ecological field surveys. Weighting factors are set to 1 by default (first column). After calculation of one effect value per line of evidence, the data are recollected in a final set Triad data in order to judge the level of (dis)agreement between the lines of evidence (lowest tabular square). When the deviation factor (D = 1.73 × standard deviation) between the lines of evidence is low enough (see text), an integrated risk value can be used for underpinning the site management decision. In the Netherlands this lay out of the table is proposed for presenting results of a Triad as part of an ERA (Mesman et al. 2007) all data, an interdisciplinary weighting over all three lines of evidence has to be applied, which has serious disadvantages. It may be argued that as well the integration within (intra) and between (inter) the various lines of evidence in principle concerns 'comparing apples and oranges'. However, for the moment it is the best approach available, although it is still open for improvement and adjustment.

The first integration process, i.e. within one line of evidence, aims to get a sufficient and complete set of information for estimating the risk from soil contamination. Different pieces of information are used for this evaluation. For instance, the application of Species Sensitivity Distributions (SSD) adopts the reasoning that all organisms are equally important, although they have a different sensitivity towards the contaminants at the site (Posthuma et al. 2002). Furthermore, estimates of effects based on contrasting exposure scenarios, like pore water and food exposure, may be used together to account for species-specific differences in bioavailability.

Table 15.2 Example on how to interpret the outcome of the integrated risk analyses of the Triad. It is highly recommended that stakeholders and risk assessors produce such a table before the start of the Triad process (reproduced with slight modifications from Jensen and Mesman 2006)

		Conclusion (land uses)	
Deviation (*D*)	Integrated risk (IR)	Acceptable	Not Acceptable
$D < 0.4$*	$0 < IR < 0.25$*	All land uses	–
	$0.25 < IR < 0.50$	A, R, I	N, A (with ecological and nature targets)
	$0.5 < IR < 0.75$	I, (R)	N, A, R (with ecological and green functions)
	$0.75 < IR < 1.0$	I (only with sealed soils)	N, A, R, I (with ecological and green functions)
$D > 0.4$ further investigations or (alternatively):	$0 < IR < 0.25$	A, R, I	N, A (with ecological and nature targets)
	$0.25 < IR < 0.50$	I, (R)	N, A, R (with ecological and green functions)
	$0.5 < IR < 1$	I (only with sealed soils)	N, A, R, I (with ecological and green functions)

*These numbers are arbitrarily chosen, and can be part of the negotiation process between stakeholders, authorities and risk assessors. The goal of this table is to demonstrate the common sense of choosing criteria for interpreting Triad results in the decision-making process.

D is a deviation factor indicating the level of disagreement between the lines of evidence of the Triad ($D = 1.73 \times$ standard deviation). IR is the integrated risk value from three different lines of evidence (arithmetic mean). 'Not acceptable' land use does not necessarily have to imply remediation or adapted soil management, but could also lead to more investigations. *N* nature, *A* agricultural sites, *R* residential sites, *I* industrial sites.

In the second and last integration step, the independent pieces of information from the three lines of evidence are compared. In this step, it is also evaluated to what extent the three lines of evidence indicate comparable levels of risks. At this point, a weight of evidence approach will pay off. Consequently, when all lines of evidence point in the same direction, it is defendable to calculate a final risk index based on the outcome of three different lines of evidence, and then compare the result with a benchmark value to be able to take a decision about the site-specific ecological risks. The benchmark value is a decided value by the stakeholders, the local administration and national government, which marks the border between acceptable and unacceptable effects (see Fig 15.1: step 5d). When the three different lines of evidence do not point in the same direction, the deviation between the three lines of evidence should be calculated and used to decide whether more research is necessary. Jensen and Mesman (2006) and Mesman et al. (2007) developed decision tables in order to arrive at these 'go/no-go decision points' to further harness a Triad approach (Table 15.2).

15.6 Embedding ERA in Formal Assessment Frameworks

15.6.1 An Example of a General Framework from the Netherlands

In the Netherlands, the Soil Protection Act was introduced in 1986. Contaminated sites are first assessed using a set of Soil Quality Standards, i.e. *Target Values* and *Intervention Values*. These values take both human health and ecological risks into account, and are applied to all kind of land uses and soil types (Rutgers and Den Besten 2005; Swartjes 1999). Recently, also so-called *Maximum Values* were introduced as remediation objectives, which are land use specific (Dirven-Van Breemen et al. 2007, 2008). The ecological basis of these Soil Quality Standards is a SSD, constructed from No-Observed Effect Concentrations (NOEC values) from the literature (Posthuma et al. 2002). At seriously contaminated sites, remediation or other soil management decisions are required if unacceptable risks cannot be refuted, based on a site-specific ecological Risk Assessment, a Human Health Risk Assessment, and the chance for dispersion of the contaminants. For these three issues, a tiered approach called the Remediation Criterion is used (VROM 2008). The first and second tiers of the ERA in the Remediation Criterion are based on a judgment of the likely ecological effects from chemical concentrations in generalized models for toxicity and mixture effects. Note that this numbering of tiers is formal and does not include the numbering of tiers in a Triad approach. In the first tier of the Remediation Criterion, the Intervention Values are used as Soil Quality Standards, besides criteria for impacted soil volume. In the second tier, ERA is performed on the basis of a calculation of the Toxic Pressure of the mixture of contaminants and a decision table addressing critical dimensions of the impacted area (Table 15.3) and presumed land use sensitivity for contamination. For a few cases, the outcome might not at all be satisfactory and sufficiently robust for a decision

Table 15.3 Scheme for supporting ERA with respect to determining the urgency of remediation at seriously contaminated sites in the Netherlands (VROM 2008, modified and currently under discussion). Depending on the land use, it is not necessary to take measures when the horizontal dimensions of the unpaved contaminated area within the contour for the Toxic Pressure (TP) is smaller than indicated. Two levels for the TP are used, i.e. TP = 0.2 or TP = 0.5

Land use	*Unpaved surface area contamination* (TP_{MMec50}* > 0.2)	*Unpaved surface area contamination* (TP_{MMec50}* > 0.5)
Sensitive: nature (including Ecological Main Structure and Natura 2000 areas)	< 500 m^2	< 50 m^2
Intermediate: agriculture, (vegetable) garden, green areas with 'nature values'**	< 5.000 m^2	< 500 m^2
Insensitive: other green area's, strips in build areas, infrastructure and industry	< 50.000 m^2	< 5.000 m^2

*TP_{MMec50} is the Toxic Pressure, which is calculated from the mixture of contaminants in soil samples (Box 15.1). The TP is calculated on the basis of total concentrations in the samples, and related to EC50 data from the literature and a mixed model for mixture effects (De Zwart and Posthuma 2005). Background concentrations of substances are subtracted from the soil sample concentrations. All concentrations are corrected for a standard soil (see Swartjes 2010). More details about the calculations can be found in Rutgers et al. (2008b).
**Outside the Ecological Main Structure and Natura 2000 areas 'nature values' are considered relevant, unless particularly stated in the petition on the land use.

about the land management regarding the contamination. In these cases, additional effort via application of the Triad approach in a subsequent tier to further improving ERA is recommended. For application of the Triad practical guidance is available (Jensen and Mesman 2006; Mesman et al. 2007; Rutgers and Den Besten 2005).

15.6.2 Examples of the Lines of Evidence in the Dutch Remediation Criterion

In the Netherlands, a practical Triad approach has been developed (Mesman et al. 2007). In Box 15.1 some examples of methods and calculation tools are presented. The three lines of evidence are described below, together with the collection of basic tools recommended for this tier of the Remediation Criterion (VROM 2008):

1. Chemistry: Model and parameters are equivalent to those at tier 2 of the Remediation Criterion (Rutgers et al. 2008b; VROM 2008). It consists of a calculation of the Toxic Pressure from the mixture of contaminants in soil samples from the contaminated site. The two-step mixed-model approach for mixture toxicity is used, combining concentration-additivity and response-additivity models (De Zwart and Posthuma 2005). In order to arrive at a number for the Toxic

Pressure which is realistic for seriously contaminated sites in the Netherlands and useful for the Triad approach, EC (Effect Concentration) data from the literature have been used (EC50 values, concentration of toxicant demonstrating 50% effect) instead of NOEC (No-Observed Effect Concentrations) data (Rutgers et al. 2008b). This is a more realistic and less conservative procedure and compatible with scaling procedures in the other lines of evidence of the Triad. A correction for bioavailability, however, is not recommended at this stage, because frameworks for implementation of bioavailability are still in development (Brand et al. 2009). Some contaminated sites, however, were assessed on the basis of testing the pore water concentrations, using basic assumptions for ERA in surface waters (Jensen and Mesman 2006; Rutgers and Den Besten 2005).

2. Bioassays: Screening level and standardized bioassays are recommended, but there is no detailed prescription. Relevant aspects for selecting a bioassay are sensitivity and validity, which should be generally accepted or carefully addressed. It is not always necessary to select bioassays with autochthonous organisms or to invest a lot of effort in this kind of tests, because differences between autochthonous and exotic organisms are usually much smaller than the differences between exposure conditions in the field and laboratory and other lab-to-field extrapolation issues (Rutgers and Den Besten 2005). Among the more popular screening tests are elutriate-based bioassays with small invertebrates, algae, plants or bacteria (e.g. *Microtox*) and whole-soil bioassay with soil invertebrates or plants. The response of the bioassays is simply expressed as a fraction of effect, ranging from 0 (no effect) to 1 (maximum theoretical effect level). More details can be found in the decision support system by Jensen and Mesman (2006).
3. Ecological observations in the field: At this stage of the Triad approach it is recommended to include plant surveys of the contaminated site and reference site(s) (with no or low contamination levels). As an alternative, simple determination of the community composition and abundance of soil invertebrates like nematodes, enchytraeids (pot-worms), earthworms and springtails may be feasible. Again, the response to the contamination should be expressed as a fraction of effect ranging from 0 (no effect) to 1 (maximum theoretical effect level). More details about the calculation of risk from multivariate observations can be found in, for example, Jensen and Mesman (2006) and Dagnino et al. (2008).

Box 15.1 Chemical characterization of effects

The Toxic Pressure (TP) from the complete mixture of contaminants in soil samples is obtained from mixture modeling using models for concentration addition (CA) and response addition (RA) (De Zwart and Posthuma 2005; Rutgers et al. 2008b). In the first steps the combination toxicity of any group with 1 or more toxicants with a comparable mode of action is calculated using the CA model:

$$\mathrm{HU_j} = \frac{[n_1]}{10^{\alpha_1}} + \frac{[n_2]}{10^{\alpha_2}} + \ldots + \frac{[n_n]}{10^{\alpha_n}} = \sum_n \frac{[n]}{10^{\alpha_n}} \text{ and: } \mathrm{TP_{CAj}} = \frac{1}{1 + e^{-\log(\mathrm{HU_j})\cdot\beta_j^{-1}}}$$

with: HU_j is the Hazard Units of the group of toxicants for which the CA model is valid. $[n_1]$, $[n_2]$, et cetera are concentrations of toxicants 1, 2,.... (e.g. in mg/kg_{dw}; after correction for a standard soil and background concentrations; see Swartjes (2010)). α_n is a log-transformed value for toxicity (e.g. a logHC50). β_j is a slope parameter of the SSD (these α_n and β_j constants can for instance be found in Rutgers et al. 2008b).

In the next step the TP of the complete mixture of toxicants is calculated using the RA model:

$$\mathrm{TP_{MM}} = 1-(1-\mathrm{TP_{CA1}})\cdot(1-\mathrm{TP_{CA2}})\ldots\ldots(1-\mathrm{TP_{CAn}}) = 1-\Pi(1-\mathrm{TP_{CAn}})$$

The Toxic Pressure obtained from the mixed model (TP_{MM}) is expressed as a multi substance Potentially Affected Fraction (msPAF) value and ranges from 0 (no effects) to 1 (theoretical maximum effect value).

Toxicity Characterization with Bioassays

The scaling of results from bioassays is usually straightforward, when test performance in control and reference samples is known. Sometimes, it is necessary to define a theoretical value for the full effect (1 = 100%). The final result can then be expressed as a fraction, ranging from 0 to 1. Examples of using and scaling results from bioassays can be found in e.g. Jensen and Mesman (2006) and Semenzin et al. (2008). The basic principles can be illustrated with earthworm tests. Data from the survival or reproduction of earthworms in contaminated and reference samples are straightforwardly fed into the ERA (ISO 16387:2004, ISO 11268-2:1998). The reference is set to 0; no survival is set to 1. The percentage of survival compared to the reference can be directly used as an effect value. The results from the chronic reproduction test can also follow this scheme, although arguments to use a different scale can be put forward. It becomes a bit more difficult with, for instance, the earthworm avoidance test (ISO 17512-1:2008). Typically, the distribution of worms between control and contaminated soil can be used on an effect scale (Amorim et al. 2005):

$$\text{Effect} = (R - C)\cdot(R + C)^{-1}$$

with: R is the number of worms in the reference or control soil; C is the number of worms in the contaminated soil. A negative outcome indicates attraction to the contaminated soil, which should be set to zero. Also with the avoidance test, arguments can be raised to use a different scaling, since avoidance is not straightforwardly controlling earthworm populations in the field.

Approximation of Effects from Ecological Field Monitoring

The scaling of single variables from ecological field monitoring can follow the same principles as bioassays (see above for an example). Typically, field monitoring deals with multiple variables, for which the scaling issue is less clear. The BKX-Triad algorithm to scale the results from multiple variables is very simple and robust, and can be used for virtually any dataset (Jensen and Mesman 2006), but has some drawbacks from unintended amplification in the effect calculation when many variables are present:

$$\text{BKX-Triad} = \text{Effect} = 1 - 10^{\sum_{i=1 \to n} \left|\log(x_j) \cdot n^{-1}\right|}$$

With x the result of the observation i divided by the result from the reference observation and n is the number of observations at the site (or in samples).

More sophisticated scaling is possible like the use of distance values in multivariate space (e.g. Euclidean distance). This is a solution to the problem of the BKX-Triad, i.e. all deviations from the references are adding to the total calculated effect. Software tools for multi-criteria analysis are easily available, but some training is necessary to use and interpret the data:

$$\text{ED} = \sqrt{\sum_k (y_{ki} - y_{kj})^2} \text{ and: Effect} = \text{ED}_{\text{R-C}} \cdot \left(\text{ED}_{\text{R-C}} + \text{ED}_{\text{C - C}theor}\right)^{-1}$$

With ED is the Euclidian distance between site (or sample) i and j for k dimensions. Subscripts R, C and C*theor* denote the reference, contaminated and theoretically contaminated site to a 100% field effect.

After a proper scaling, the outcome of different lines of evidence should be in balance. This balance will be theoretically demonstrated with a very large number of tests addressing the set of end-points within each line of evidence. With a subset of tests, like the test proposed in the Dutch Triad approach summarized above, deviations from this balance should be expected and interpreted in terms of model uncertainties. However, if the outcome of a subset already demonstrates convergence of the results, then this is a strong basis for finalization of the ERA, providing a solid advice for the Risk Assessment or Risk Management of the site. As a practical criterion for convergence, the deviation between the outcomes of different Triad approach lines of evidence can be quantified as suggested by Jensen and Mesman (2006) and listed in Table 15.1.

15.6.3 Outline of ERA in Other Countries

The United Kingdom has developed a framework for ERA (Weeks and Comber 2005; Weeks et al. 2004). A cornerstone in this framework is the connection to the statutory regime for identification and control of sites potentially affected by contamination, also known as Part IIA of the Environmental Protection Act of 1990. This act defines a site 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;
- contamination of controlled waters is occurring or is likely to occur.

Currently, only ecological risks to controlled waters and certain protected habitats (defined in Part IIA) are covered. The framework, however, does address how to perform ERA at sites not currently covered by Part IIA. The UK framework is based on schemes found in e.g. USA, Canada and the Netherlands. Like the procedures in these countries, it is a based on a tiered approach, where the initial tier zero aims to determine whether or not a site belongs under the Part IIA of the legislation. It involves the development of a Conceptual Site Model, which describes what is already (historically) known about the site, e.g. whether there is a likely source-pathway-receptor linkage.

In many other countries of the EU, for example Germany, Spain and Sweden, ERA can be based on additional types of testing, making a Triad approach framework feasible. A decision support system for assisting in site-specific ERA was developed based on research at the Acna di Cengio 'mega site' in the Bormida valley, Italy (Critto et al. 2007; Semenzin et al. 2007, 2008, 2009a, b).

The US and Canada were among the first in producing general frameworks for ERA (CCME 1996; US-EPA 1998). Later many amendments to the first publications were produced and these are available via the respective websites.[1] Both US and Canada frameworks for ERA address many questions related to relatively large contaminated areas, whereas some European approaches typically are designed to cope with many but smaller sites. Furthermore, wild life is a more important issue in the North America frameworks compared to the European. The reason for the somewhat reversed development of ERA in the two regions might be due to the fact that Soil Quality Standards were first developed in Europe, while general frameworks were first developed in North America. Nowadays the basic outlines of the various ERA frameworks and derivation of Soil Quality Standards world-wide seem to converge (Swartjes et al. 2008).

[1] http://www.epa.gov/riskassessment/ecological-risk.htm; http://www.ccme.ca/ourwork/soil.html

15.7 Outlook

In the Dutch Soil Protection Act the ecosystem is relevant for any kind of land use, although the ecological protection level varies (e.g. nature is considered more sensitive than industrial land use; Rutgers et al. (2008b); Swartjes (1999)). This has triggered attention to ERA, principally at all contaminated sites, also outside nature areas. Public support of this policy is limited, especially in cases where there is no visible damage to the terrestrial ecosystem. This has to do with lack of knowledge and, hence, appreciation for the tasks and significance of the soil ecosystem, among the general public (see Chapter 13 by Swartjes et al., this book). Results from a Triad approach-based assessment will support the acceptation of remediation measures by the general public in these cases. Acceptation is expected to improve further from increased environmental awareness due to climate change and rephrasing soil functions into the goods and services of the soil system (Rutgers et al. 2009).

A compelling recommendation to use the Triad approach in the higher tiers of ERA will result in increased attention on ecological issues and habitat protection. This was observed in the Netherlands (SKB 2009). In the lower tiers of ERA (tier one and two in the remediation criterion; Rutgers et al. (2008b)), the surface area exceeding a threshold for Toxic Pressure of the mixture of contaminants has to be remediated. In many cases in the Netherlands, the goal of a Triad approach-based ERA in tier three and subsequent tiers is then to reduce the surface area to be remediated and hence to reduce costs. Step by step, the Dutch regulators have become less hesitant with respect to interpreting Triad approach-based results. This was concluded from an inventory from 42 Triad approach-based assessments with an evaluation of the interpretation and integration of the results, and the decision-making process (SKB 2009). In 63% of the cases (total 45: the unknowns were omitted from the analysis), the management and remediation decisions were adjusted in reaction to the results from the Triad-based assessments. Since local administrations (e.g. provinces) and the national government have ratified an agreement on speeding up the soil remediation (Covenant 2009), and the procedural standard for a guidance on incorporation of a Triad in ERA will be soon available (NEN 2010), it is expected that the number of Triad-based Risk Assessments will further increase.

Although many tools for a Triad approach in ERA are available, there is still a strong demand for improved and robust methods in many cases. Also, many methods are considered not cost-effective or too laborious for smaller cases. Consequently, increasing the number of Triad-based Risk Assessments will demand for improved, new, standardized, robust and cost-effective tools.

Although ecological surveys in principle are the most site-specific part of an ERA, it is often hampered by a weak relation between contamination levels in soil and ecological observations and, hence, may lack plausibility. Ecosystems, communities and populations of organisms are shaped by a comprehensive set of environmental factors, where soil contamination is only one of those factors. Furthermore, ecological field observations occasionally need highly-trained experts and a relatively large effort.

So called ‘omic-’techniques, like genomics based on analysis of DNA-patterns, were advocated for a wider application in ERA, but progress is limited yet. The promise is that these techniques generate much and valuable information with a limited effort. However, the methods are generally immature and still quite expensive compared to traditional bioassays and community analysis. Furthermore, the issue of interpretation of community shifts for ERA is not fully resolved, i.e. scaling is still an issue. Nevertheless there is still hope for breakthroughs in this area thanks to expected technical spin-offs from medical research and agriculture business.

Further stimulating influences can be expected from international harmonization of models and frameworks for ERA by e.g. the HERACLES network (acronym: human and ecological Risk Assessment for contaminated land in European member states; Swartjes et al. (2008)). The conclusion from the network was that quite a number of Member States had readily available tools for implementing ERA. In the nearest future the Habitat and the Water Framework Directives are most likely the dominant drivers in introducing ERA in some form to a wider number of countries. In addition, also the EU soil thematic strategy and further elaborations into the future Soil Framework Directive will further stimulate attention to ERA at contaminated sites. The Triad approach will be part of these developments, triggering further improvements to previously addressed issues and new developments.

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