Jaana Sorvari

Application of Risk Assessment and Multi-Criteria Analysis in Contaminated Land Management in Finland

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Jaana Sorvari

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List of original publications

This dissertation is based on the work presented in the following original papers, which are referred to using Roman numerals.


The author’s contribution to the above-mentioned publications and the studies involved is as follows:

I J. Sorvari wrote the paper and planned, conducted, and interpreted all calculations related to the TRIAD procedure, statistical analysis, MCA, and the ecological risk assessment based on chemical studies.

II J. Sorvari was in charge of writing the paper, planned and conducted the risk assessment, and participated in the planning of site studies.

III J. Sorvari was in charge of writing the paper, planning and implementing the study, and constructing the DST; she conducted all calculations, designed and constructed the three DST modules, and participated in the construction of the remaining two modules.

IV J. Sorvari is the principal author of the paper, was in charge of planning, supervised the implementation of the study, conducted the risk assessment part and the survey of the decision support systems and tools, and participated in other parts of the study.

V J. Sorvari is the sole author of the paper and she planned, conducted, and interpreted the risk assessments.

VI J. Sorvari reviewed and commented the manuscript and participated in the risk assessment, while the other authors were in charge of implementing the studies and preparing the paper.
Application of risk assessment and multi-criteria analysis in contaminated land management in Finland.

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Abstract

Land contamination is a significant environmental problem requiring systematic management actions. Defining the type and scale of the actions requires information on the risks involved. The numerous methods available for conducting risk assessment (RA) vary in terms of complexity, level of detail, conservatism, and outcomes. Thus, selecting suitable methods requires information on their applicability in Finnish conditions and at the specific site. On the other hand, it is generally accepted that current contaminated land management (CLM) should not only focus on minimizing site-specific risks, but should also consider overall environmental effects and socio-cultural and socio-economic aspects. Multi-Criteria Analysis (MCA) could then be used as a tool for integrating multidimensional data and generating aggregated information on the consequences of different risk management (RM) options, such as environmental, social, and economic impacts. Nonetheless, such approaches have very seldom been applied in CLM in Finland, probably partly due to a lack of tools specifically developed or modified for Finnish conditions.

This research studied the application and suitability of different RA methods for assessing risks and identifying RM needs at some typical contaminated sites in Finland and demonstrated the use of MCA, the emphasis being on soil contamination. The studied RA approaches comprised qualitative rating and quantitative methods that were based on using environmental benchmarks, uptake and exposure models, and multimedia software. To derive estimates of ecological risks, the so-called TRIAD procedure that uses chemical studies, bioassays, and ecological studies was also applied and combined with MCA in order to account for the performance of the study methods, i.e. their ability to depict ecological risks at a study site. Qualitative rating and the statistical Monte Carlo technique provided additional means for uncertainty analysis. A separate study applying the Metaplan technique, interviews, a questionnaire, and a literature survey showed that a lack of suitable assessment tools was one of the key barriers to eco-efficient CLM in Finland. An MCA-based decision support tool (DST) adapting the Multi-Attribute Value Theory (MAVT) was therefore developed for case-by-case determination of the preferred RM option and tested with some typical Finnish contaminated sites.

Many of the conclusions of the research are overarching and applicable to RA methods in general. Firstly, it appered that care must be taken in applying different models and software tools in site-specific RA, since some of their components are not straightforwardly suitable for Finnish conditions or for certain contaminants. These problems often relate to specific contaminant transport pathways. Moreover, the lack of verified data on the parameter values representative of Finnish conditions is an issue. The prevailing practice of using complicated software programs with ample data demands as the first and primary tools in human health risk assessment is not supported by this research, since it appeared that even simple tools and calculations can often
provide adequate information on risks for decision-making. In ecological risk assessment (ERA), the usefulness of the approach founded on uptake and exposure models is reduced by the high uncertainties involved, particularly since the applicability of these models in Finnish conditions could not be verified. The accuracy and reliability of ecological risk estimates can be enhanced by applying the TRIAD methodology, although the procedure includes some pitfalls that need to be acknowledged. Combining TRIAD with MCA proved to be a feasible means to quantitatively study the performance of separate ERA methods. MCA thereby complements mechanical statistical analysis, such as Monte Carlo simulation, and increases the reliability of the final integrated risk estimates. In practice, a lack of data on the statistics of the input variables can restrict the use of statistical tools. The MAVT-based DST turned out to be efficient in facilitating discussion between different interest groups and experts and in identifying the preferred RM option in the common situation where risks are not the only factors relevant in decision-making. In practice, additional factors, such as the temporal scope of RM actions and some sustainability components that were not comprehensively included in the DST, might need to be considered.

*Keywords:* contamination, soil, groundwater, risk assessment, risk management, decision support tool, health risk, ecological risk
Riskinarviointimenetelmien ja monikriteerianalyysin soveltaminen pilaantuneiden maa-alueiden riskinhallinnassa Suomessa

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Tiivistelmä


Suurin osa tutkimuksen johtopäätöksestä pätee yleisesti riskinarviointimenetelmiin. Tutkimuksessa ilmeni ensinnäkin, että erilaisia malleja ja laskentatoimintoja tulisi käyttää harkitsevästi, sillä kaikki niihin sisältyvät arviointiosiot eivät sovellu suoraan Suomen olosuhteisiin tai tiettyjen haittaaineiden arviointiin. Nämä rajoitteet tulevat usein esiin tietyillä kulkeutumisreiteillä. Tiedonpuute

Aisiasanat: pilaantuminen, maaperä, pohjavesi, riskinarviointi, riskinhallinta, riskien hallinta, päätöksenteko, monikriteerianalyysi, terveysriski, ekologinen riski
Terms and acronyms

AHP Analytical hierarchy procedure; one of the MCDA techniques (see below)
BAT Best available technology
Benchmark A cutoff value for contaminant concentration: exceedance indicates risks above a set protection level, based on, for example, protecting 95% of all organisms as defined statistically or a no-observed-effect concentration (NOEC) value of a particular organism
BEP Best environmental practice
Bioavailability The degree to which a substance becomes available to the target tissue after administration or exposure (IRIS, 2010)
BKM Bio-kinetic model
CBA Cost-benefit analysis
CEA Cost-effectiveness analysis
CLM Contaminated land management; at a site level this covers any management practices and actions for restricting the unwanted consequences caused by soil or groundwater contamination (see also RM)
CM Conceptual model
COP(E)C Contaminant of potential (ecological) concern
Dose Total amount of a substance administered to, taken up, or absorbed by an organism, organ, or tissue
DSS Decision support system
DST Decision support tool
ERA Ecological risk assessment
Eco-efficiency Indicates the ratio of net environmental benefits to the resources expended and any negative effects associated with risk management actions; environmental effects are understood to include social and socio-cultural aspects related to the quality of life.
Exposure Process by which a substance becomes available for absorption by the target population, organism, organ, tissue, or cell, by any route
Hormesis A stimulatory effect in a living organism, which can occur, for example, as increased growth in concentration levels above the control (i.e. uncontaminated medium)
HRA Health risk assessment
In vivo In the living organism
In vitro In an artificial environment outside the living organism
LCA Life cycle analysis
MAUT/MAVT Multi-attribute utility/value theory, one of the MC(D)A techniques (see below)
MCA/MCDA Multi-criteria analysis, multi-criteria decision analysis/aid (synonymous to MCA); a discipline that covers various methods aimed at supporting decision makers in solving complex and multidimensional decision problems
MNA Monitored natural attenuation
Multimedia model A model that covers several environmental compartments (e.g. soil, groundwater, surface water, air)
<table>
<thead>
<tr>
<th>Acronym</th>
<th>Description</th>
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<tbody>
<tr>
<td>NAPL, LNAPL</td>
<td>Non-aqueous phase liquid, light non-aqueous phase liquid</td>
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<td>NICOLE</td>
<td>Network for Industrially Contaminated Land in Europe (1998—ongoing)</td>
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<tr>
<td>NOEC/NOEL</td>
<td>No-observed-effect concentration/level; highest concentration or amount of a substance, found by experiment or observation, that causes no specified adverse effects in the target organism</td>
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<tr>
<td>PAH</td>
<td>Polycyclic aromatic hydrocarbon</td>
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<tr>
<td>PHC</td>
<td>Petroleum hydrocarbon</td>
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<tr>
<td>PCB</td>
<td>Polychlorinated biphenyl</td>
</tr>
<tr>
<td>PCDD, PCDF</td>
<td>Polychlorinated dibenzo-p-dioxin, polychlorinated dibenzofuran</td>
</tr>
<tr>
<td>Point of compliance (POC)</td>
<td>The location in the environmental compartment (e.g. soil, groundwater) where the contaminant concentration should not exceed the set acceptable value (e.g. drinking water guideline, soil quality standard)</td>
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<tr>
<td>QSAR</td>
<td>Quantitative structure activity relationship; QSAR model is a mathematical equation which determines a relationship between chemical structure and behavior of chemicals, such as chemical reactivity or biological activity</td>
</tr>
<tr>
<td>RA</td>
<td>Risk assessment</td>
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<tr>
<td>RBCA</td>
<td>Risk Based Corrective Action</td>
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<tr>
<td>RBLM</td>
<td>Risk-based land management</td>
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<tr>
<td>Reference value</td>
<td>A value used in the determination of health risk estimates, such as acceptable daily dose (ADI) or cancer slope factor</td>
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<td>RM</td>
<td>Risk management; at the site-specific level RM covers any actions to control risks, including active remediation measures, monitoring, isolation of the area and land use restrictions</td>
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<tr>
<td>SAMASE</td>
<td>National project on the research and remediation of contaminated soils (1989—1994)</td>
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<tr>
<td>SSV</td>
<td>Soil screening value; a soil quality benchmark applied in decision-making</td>
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<tr>
<td>TPH</td>
<td>Total petroleum hydrocarbon</td>
</tr>
<tr>
<td>Uptake</td>
<td>Entry of a substance into the body, into an organ, into a tissue, into a cell, or into the body fluids by passage through a membrane or by other means</td>
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1 Introduction

1.1 Contaminated sites in Finland

Land contamination is a significant environmental problem recognized worldwide. In Finland, the problem was initially revealed in the national SAMASE project carried out by the environmental administration between 1989 and 1994. SAMASE identified altogether 10,000 sites with potential soil and/or groundwater contamination (Puolanne et al., 1994). The survey has since been updated, and the current figure totals about 21,000 (Finnish Environment Institute, 2009a). Gasoline stations form one-third of these sites. Other major activities that have created contaminated sites include repair shops, paint shops, and scrap yards (14%); landfills (11%); saw mills and impregnation plants (6%); and shooting ranges. Of all the contaminated sites, some 4,000 are located on groundwater catchment areas. Approximately 4,000 contaminated sites have been remediated during the last 20 years, while the number of annual remedial decisions currently totals some 300 cases (Finnish Environment Institute, 2009b), the corresponding costs of remediation being 50-70 Meuro per year (Pajukallio, 2006).

The primary initiator of investigations and remediation at potentially contaminated sites in Finland is a change in land use, while human health risks or risks to other recipients are the major causes in some 20% of contamination cases (Finnish Environment Institute, 2009b). Other major initiators include construction work, whereas administrative impositions appear in only some 1% of all remediation projects.

1.2 Framework for decision-making

Decisions on contaminated land management (CLM) actions have traditionally been based on the contamination level. The risks to human health or the environment associated with the contamination can be either perceived or actual. In the beginning of the history of land remediation the problem was generally managed by aiming at a “zero risk level”, often using natural background levels as remedial targets (e.g. Nathanail and Bardos, 2004). However, this approach soon appeared unfeasible due to the extensive remediation measures and unbearable costs involved. This led to the adoption of the Risk-Based Land Management (RBLM) approach where the actual risks rather than rigid concentrations drive the CLM decision. Nowadays it is widely acknowledged that in practice, additional drivers are involved in decision-making. CLM is in fact a multi-dimensional decision problem involving several contributing factors (Fig. 1) and hence, a transdisciplinary and holistic approach is needed in dealing with it. At present the focus is particularly on “green remediation”, i.e. remediation which aims at maximizing environmental benefits and thereby reaching sustainability. This involves consideration of all material and energy flows and long-term impacts on ecosystems and human health.

Although risks are the primary drivers of risk management (RM) actions, available resources, particularly time and money, play a significant role in the selection of RM methods in Finland (Sorvari, 2004a & 2005b; Sorvari and Antikainen, 2004). The poor availability and feasibility of some remediation techniques can also limit the choice of RM alternatives (see Section 1.3.). In some cases socio-cultural aspects, such as preserving heritage buildings, can hinder the use of

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1 RBLM can also be understood in a broad sense as meaning a general strategy that covers the integration of approaches originating from different perspectives, such as spatial planning, environmental protection, and engineering (Vegter et al., 2003). The goals of such RBLM includes comparable levels of protection of health and the environment, taking into account local characteristics; optimized use and development of technical and administrative solutions; and sustainability (evaluating and optimizing environmental, economic, and social factors).
invasive remediation techniques, while sometimes maintaining a positive public image or sustaining the attractiveness of the site can promote their use.

Decision support systems (DSS) and decision support tools (DST) can assist solving complex decision problems, such as risk management of contaminated sites. It needs to be mentioned that the definitions of DSS and DST vary in different sources. While some studies define DSSs as “computer-based systems that facilitate the use of data, models, and structured decision processes in decision-making” (Sullivan et al., 2007), others consider a DSS something that “guides risk assessors through an assessment, for example the assessment of the soil quality, according to a fixed procedure” (Swartjes et al., 2009), the latter definition being broader. According to Swartjes et al. (2009), DSSs usually also involve policy aspects. Similarly, DSTs can be understood to only include quantitative tools, but alternatively in a wider meaning, to also cover qualitative documents (e.g. Onwubuya et al., 2009) “produced with the aim of supporting decision-making, i.e., something that carries out a process in decision-making” (Bardos et al., 2003). In this research, a DST is understood to be any quantitative tool that supports decision-making, and is used for conducting risk assessment, cost-benefit analysis, environmental impact assessment, life cycle analysis, and sustainability or eco-efficiency analysis. DSSs again are considered to also include qualitative procedures, written guidance, and guidelines, i.e. any methods that can guide a decision-maker. DSTs also cover tools for decision-making when several incompatible criteria, such as risks, costs, and environmental effects, are involved. Paper III briefly describes such a multi-criteria DST. This Finnish DST is one of the key components of the DSS developed in the PIRRE project (Sorvari, 2004a; 2005b).

The multidisciplinary character of CLM also means that there are usually several parties involved in the decision-making process. Involving all the relevant stakeholders in decision-making and communicating the risks is a challenge recognized worldwide (e.g. Petts et al., 2003; Schewald-van der Kley, 2004). DSSs and DSTs planned for group decision-making, which involves several stakeholders, provide a means to communicate and discuss possible conflicting views and preferences in such a situation in order to reach a consensus.

![Figure 1. Framework for decision-making regarding contaminated sites.](https://www.environment.fi/syke/pirre)
1.3 Risk management practices in Finland

Environmental legislation provides the foundation for all decision-making in CLM. Current environmental legislation in Finland includes both generic principles of assessing the potential risks involved and impositions and guidelines, such as environmental standards to be used in the definition of remediation need.

Selection of RM measures for contaminated land largely depends on the type and scale of contamination as well as on the environmental characteristics of a specific site, since these factors determine the magnitude of risks and which RM methods are applicable. Environmental conditions, e.g. cold climate and challenging hydrogeochemical conditions (e.g. heterogeneous soils and fractured bedrock), make some remediation methods unfeasible in Finland (e.g. Penttinen, 2001). They also complicate assessment of the time span of risk reduction in the case of in situ methods such as monitored natural attenuation (MNA) (Jørgensen et al., 2006) and consequently, planning of RM actions.

The Finnish environmental legislation defines the generic principles to be followed in activities that could pollute the environment, such as the Best Environmental Practice (BEP) and Best Available Technology (BAT) principles (YSL, 2000). These also apply to any RM actions at contaminated sites. While guidance in complying with the BEP principle already exists (Mroueh et al., 2004), generic criteria for BAT-accordant CLM are only in preparation (Outi Pyy, personal communication, 29 Dec 2009). The latter guidelines will be accompanied by an Excel-based assessment tool, which can be used for preliminary screening of remediation techniques, the feasibility of which could be further evaluated using a suitable DST. A survey conducted within the PIRRE project showed that such guidelines and tools are evidently needed to enhance eco-efficiency and sustainability in CLM (Sorvari and Antikainen, 2004, Appendix 3; Sorvari, 2005a & b).

Excavation combined with disposal or ex situ treatment is still the predominant soil remediation method in Finland, since more than 90% of previous remediation projects have been mostly based on using this traditional methodology (Finnish Environment Institute, 2009b). Active cleaning operations to remediate groundwater have seldom been carried out, but it seems that some in situ methods such as MNA and reactive walls are being increasingly adopted3. In situ methods have been used most extensively in the remediation of former gasoline stations (Nikunen, 2010).

The superiority of soil excavation in Finland and many other European countries (Reinikainen, 2009) is generally based on the fact that it is a quick and the most reliable method for reducing risks, leaving no future liabilities at the contaminated site. It is however, widely recognized that soil excavation creates adverse environmental impacts, which reduce the sustainability of remediation. Moreover, soil replacement can also be costly compared with less invasive techniques (e.g. Lunden, 2008), although the overall costs are highly dependent on the market for excavated soil. So far, most of the excavated contaminated soils in Finland have been delivered to landfills; in 2005 and 2006 more than 80 % of these soils were utilized in different structures and daily cover (Jaakkonen, 2008). The demand for slightly contaminated soils in landfills has sometimes led to transport distances as long as 500 km (Uusimaa Regional Council, 2002). At the same time, several barriers impede recycling of contaminated (or treated) soil elsewhere (Sorvari, 2004a & b, 2005a & b).

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3 Information based on a survey (unpublished) of the decisions on the notifications and permits concerning soil remediation (available at: https://www.ymparisto.fi) as of 31 January, 2010, survey conducted by J. Sorvari
1.4 Decision support systems in contaminated land management (CLM)

1.4.1 Previous and current approaches

Recognition of land contamination problem soon resulted in issuing of guidelines on how to define remediation need and set remedial targets for a single contaminated site. In Finland such guidelines included cutoff values for concentrations of soil contaminants – known as SAMASE values (Puolanne et al., 1994), these previously being the major tools used as the starting point of risk management (Sorvari and Assmuth, 1999; Mäenpää, 2002). Although the basis of the SAMASE values was not explicitly described and they were originally meant to be used only as advisory standards, they were very often considered strict remedial targets. At the same time, the quality standards for domestic water (update in STM, 2000) were applied to aquifers even though the water was not used nor planned to be used for water supply. The SAMASE values did not have any regulatory status, whereas the new soil quality benchmarks issued in 2007 are embedded in a government decree (Ministry of the Environment, 2007). Although the new decree sets distinct numeric values for assessing remediation need, it also enhances the site-specific assessment of risks in this evaluation. The new benchmarks included in the decree were derived in accordance with the generic risk assessment (RA) methods applied in the Netherlands (Reinikainen, 2007; Sorvari and Reinikainen, 2007). They are therefore more clearly risk-based than the previous SAMASE values. The Dutch approach, documented in Baars et al. (2001), Lijzen et al. (2001), Otte et al. (2001) and Verbruggen et al. (2001), has been adopted in many European countries (Carlon, 2007).

At present it is widely recognized that in many cases generic concentration limits are not suitable or adequate for assessing risks and identifying RM needs at contaminated sites. In such cases risk assessment that considers the specific characteristics of the site and contamination is applied. In the context of CLM, RA generally only covers toxicity-associated risks, i.e. risks to human health, biota, and groundwater quality, whereas economic, social, technical, and indirect risks (e.g. to a treatment facility receiving discharge or wastes) are excluded and studied separately, when relevant.

The history of RA in the context of CLM started on a large scale in the 1970s in the USA along with the federal government’s Superfund program, which aims to clean up uncontrolled hazardous waste sites (USEPA, 2009b). At the European level, RBLM became a major topic in the European CARACAS network, and the term was established in the following CLARINET network (Vegter et al., 2003). By that time, detailed site-specific RAs had only been conducted in a few Finnish CLM projects focusing on health risk assessment (HRA) (Sorvari and Assmuth, 1999 & 2000). Moreover, these few assessments had various shortcomings. Acknowledgement of the need for specific guidance to promote risk-based CLM resulted in the preparation of the first Finnish guidelines on risk assessment at contaminated sites (Sorvari and Assmuth, 1998), followed by a guidebook on ecological risk assessment (ERA) (Pellinen et al., 2007) and generic guidelines on the application of the new soil quality benchmarks and RA in the assessment of contamination level and remediation need (Pyy et al., 2007). Despite these efforts to increase knowledge about risk assessment, more recent surveys showed that site-specific RAs still included several shortcomings and lacked transparency in documentation (Sorvari, 2004c; Hourula, 2007; Paper IV). These problems were probably mainly due to the limited resources and expertise reserved for the RA process and inadequate formulation of its targets. It also became evident that information is needed on the suitability, limitations, and data needs of different RA tools, taking into account the Finnish conditions.
1.4.2 Risk assessment (RA)

Procedure

There are various protocols as well qualitative and quantitative methods for conducting a site-specific risk assessment. The choice of methods depends on several factors such as the scale and type of contamination, the expected accuracy and reliability of the risk estimates, and available resources. Qualitative RA aims at linguistic estimates of risks, while quantitative assessment produces numeric estimates. The former method is generally used as the first stage of site-specific RA since it can be useful in identifying major risk factors, i.e. key contaminants, contaminant transport pathways, exposure routes, or relevant recipients and receptors. Qualitative ratings are also used in the ranking of multiple sites (e.g. Naumanen et al., 2002; Sorvari et al., 2006). To set specific remedial targets (concentration limits), some quantitative data on risks are generally required and therefore, quantitative methods and tools need to be applied.

Quantitative RA methods range from simple screening based on different risk-based numeric criteria, such as soil quality benchmarks, to utilization of quantitative models, bioassays, biomonitoring, and statistical tools. It has been generally recommended to follow a tiered procedure, i.e. to proceed from simple screening-level methods towards more complex methods, if warranted. In the context of CLM, such a tiered approach has been incorporated into RA frameworks in countries around the world, including Finland (Sorvari and Assmuth, 1998) and several European countries (Ferguson, 1999; Swartjes, 1999; Nathanail and Bardos, 2004; Faber, 2006), the USA (USEPA, 1996), Canada (Environment Canada, 1994), and Australia (NEPC, 1999). The reasoning behind such an approach is optimization of resources, since complex tools require more data, the compiling of which takes time and money. Moreover, screening-level or baseline RA can sometimes provide sufficient information for decision-making, thereby making a detailed assessment unnecessary. For example, a detailed quantitative probabilistic RA (see below) requires data on the statistics of the variables involved as well as their distribution and mutual correlations. Such data might not be readily available.

The RA process (Fig. 2) starts with identification of risks, followed by their determination and characterization. In the case of RA based on exposure estimates – the most frequently applied approach to assessing health risks related to contaminated sites – identification of risks requires preliminary site data on the environmental concentrations of contaminants in order to identify chemicals/contaminants of potential concern (COPCs); receptors involved and their potential exposure routes; and generic data on the physicochemical properties and toxicity of contaminants involved. These data are aggregated in a conceptual model (CM), which forms the basis for RA. CM is a schematic description of the problem, i.e. the sources of COPCs, their transport pathways, recipients, exposure routes and receptors, and hence, it defines the objectives and boundaries of RA. CM can be updated and specified along the RA process when more data become available.

Formulation of the RA problem is followed by an analysis phase, which includes determining the relevant sources, transport pathways and exposure routes of COPCs, as well as transport and exposure rates, and dose-response relationships. The final phase, risk characterization, involves aggregating all the information and describing the magnitude, spatial, and temporal dimensions and uncertainties of the risks. It is worth noting that feedback to previous work phases is usually warranted if a more detailed assessment or additional data are needed.

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1 Terms “Site Conceptual Model” (SCM) and “Conceptual Site Model” (SCM) are also used in the CLM literature.
Generation of dose estimates to determine site-specific risk estimates requires data on the concentrations of COPCs in the relevant contact or exposure media (e.g., soil, water, food, air, sediment). Models are normally used to assess contaminant transport from one environmental compartment to another when no adequate site data are available (see ‘Models and calculation tools’ section). An alternative (or complementary) approach is to conduct laboratory studies, such as solubility tests, leaching tests, and plant uptake studies. Added to these, *in situ* studies using, for example, lysimeters and air particle samplers, provide information on the mobility of COPCs in actual site conditions. Using real concentration data that covers the different media involved reduces the uncertainty of RA and is a preferable approach in higher assessment tiers.

Bioavailability of contaminants is a crucial factor in the formation of risks, and also increasingly emphasized in risk assessment of contaminated sites. Hence, several research projects generating data and methods for assessing bioavailability have been realized in Europe during recent years (e.g., Jensen, 2003). Experimental animal models have been traditionally used to study human bioavailability of lead (and other metals) in soil (e.g., Freeman *et al.* , 1992; Casteel *et al.*, 1997; Mushak, 1998). Animal models are generally considered the most reliable methods for determining bioavailability, but they are also costly and complex due to *in vivo* tests on the laboratory animals involved (NEPI, 2000). *In vitro* extraction tests provide a more feasible means to study bioavailability, and several methods have been developed to simulate contaminant fate in the human gastrointestinal tract, lead being the most frequently studied contaminant. Due to differences in test conditions, different tests produce considerably varying estimates of bioaccessibility (Oomen *et al.*, 2002). As far as is known, human bioavailability tests have not been applied in studies of

**Methods**

Figure 2. Description of the risk assessment (RA) procedure when risks are assessed on the basis of exposure estimates (adapted from USEPA, 1998). CM = conceptual model, COPEC = Contaminant of potential ecological concern.
contaminated sites has not been reported in Finland. Inclusion of generic bioavailability factors in exposure calculations instead of solubility and in vitro or in vivo tests is a feasible and suitable alternative approach in screening-level RA, but so far it has seldom been adopted in official RA frameworks (Oomen et al., 2006). In ERA, determination of (bio)availability has been based on models that assume the dependency of the environmental fate of a chemical on its structure, and empirical equilibrium partitioning coefficients (Frische et al., 2003). In addition to such Quantitative Structure Activity Relationship (QSAR) models, experimental methods including different extraction tests are being increasingly applied; in Finland these have been used particularly in assessment of phytoavailability, i.e. availability of contaminants to plants. Finally, bioavailability can be determined by measuring contaminant concentrations in site biota or in test organisms exposed to samples taken from the study site (bioassays). Novel methods to assess bioavailability include various bacterial biosensors, developed particularly for investigation of soils contaminated by metals (e.g. Turpeinen, 2002; Petänen and Romantschuk, 2003).

While potential adverse effects of non-carcinogenic chemicals on human health are usually characterized by proportioning calculated daily dose estimates to the highest doses still safe to humans (e.g. Reference Doses, RfD, Acceptable/Tolerable Daily Intakes, ADI/TDI), differences exist in the treatment of carcinogens, i.e. whether a cancer slope factor or a safe dose (e.g. ADI or Risk Specific Dose, RSD) is used to derive site-specific risk estimates. Moreover, classifications of chemicals into carcinogens and non-carcinogens vary. An alternative methodology to the application of safe doses uses biokinetic models (see ‘Models and calculation tools’ section). In the context of HRA, such models aim to describe the relationship between a specific dose and concentration in the body (e.g. in tissue or blood), thereby also taking bioavailability into account. The fundamental differences between these alternative methodologies and their manifestations in site-specific risk estimates are shown and briefly discussed in Paper V. Lastly, HRA can utilize biomonitoring or epidemiological studies to verify exposure to COPCs or to find out whether a link between contamination and health effects exists at the population or community level.

Several approaches have been applied in conducting site-specific ecological risk assessments. Quantitative ERA can be founded on studies of the biota at the study site, laboratory-scale ecotoxicity tests (bioassays), constructed model ecosystems (microcosms, mesocosms), or mathematical models describing uptake or exposure by biota, similarly to the medium–exposure–response approach used in HRA. Field studies include, for example, analyzing biomarkers or contaminant concentrations in key receptors’ tissues and ecological studies on the diversity and abundance of different species. Model ecosystems are small-scale experimental systems that endeavor to imitate the conditions (e.g. biota, soil properties) at a specific study site. In Finland, such systems were used in a project focused on phytoremediation (remediation using plants) of an industrial site contaminated by metals (Helmisaari et al., 2007). In its simplest form, quantitative (or semi-quantitative) ERA is based on a comparison of ecological benchmarks and environmental concentrations of COPECs at the study site. This approach is common in most European countries (Schewald, 2001). Although bioassays for testing soil have been under intensive development during the last few years, and they are involved in ERA guidelines in many countries (e.g. Weeks et al., 2004; Jensen and Mesman, 2006; Pellinen et al., 2007), difficulties in interpreting the results have somewhat limited their routine application (Crommentuijn et al., 2001). These difficulties arise, inter alia from the fact that toxic responses in studies based on bioassays may also reflect the presence of physical stressors, such as unsuitable soil characteristics, rather than contaminants. It has therefore been suggested that the scope of ERA should be expanded to include biological and physical stressors explicitly, i.e. to put chemical stressors in ecological context (Kaputska, 2008).

A fundamental difference in the approach to ecological risks at contaminated sites between the USA (and Canada) and European countries is worth mentioning, since this difference has its implications in ERA methods as well as in soil quality criteria and ecological benchmarks. In the
USA, the focus of ERA has traditionally been on wildlife protection (Walden, 2005a; USEPA, 2010), whereas in Europe the functionality of soil as an ecosystem is emphasized (e.g. Faber, 2006; Gardi et al., 2009). This has lead to development and use of different methods (see section ‘Models and calculation tools’). The approach known as TRIAD was recently adopted in several studies of contaminated sites in Europe (see Paper I). In the context of ERA, TRIAD\(^5\) refers to a process where the results from three different assessment methodologies, i.e. chemical studies, toxicological studies, and ecological studies, are combined in a systematic way to produce more realistic estimates of ecological risks. The different ERA methodologies involved in TRIAD are known as lines of evidence (LoE). According to Swartjes et al. (2008), the open issues in TRIAD, possibly limiting its use, include proper inclusion of bioavailability and a lack of experimental applications representative of varying environments and soil types. As far as is known, the case studies presented in Sorvari et al. (2007), Karjalainen et al. (2009) and Paper I are the only Finnish site-specific ERAs associated with soil contamination that applied the TRIAD procedure.

Notwithstanding the development of ERA procedures, methods, and tools, such as bioassays, the practices of conducting ERA are still under development in Finland as well as in many other countries. At the European level, the question of when a detailed site-specific ERA actually needs to be conducted and at what level has raised discussion (NICOLE Ecological Risk Assessment Working Group, 2006). While in the USA, combining site-specific ERA with the natural resource damage assessment (NRDA), which drives the restoration and compensation decisions at hazardous waste sites, is under consideration (Burger, 2008; Munns Jr et al., 2009). In such a combined process the focus of ERA would be on the ecological assessment endpoints in ecosystem services that are in accordance with those involved in the restoration and damage compensation decisions. The measures of biodiversity would then be the key measurement endpoints.

In practice, land contamination is seldom caused by a single chemical, and in many cases several concurrent chemicals with differing environmental fates are involved. Their combined toxic effects then become an issue. Several organizations have issued guidelines for addressing possible mixture effects associated with human exposure (USEPA, 1986; NRC, 1989; Choudhury et al., 2000; ATSDR, 2004). Non-interactive contaminants are generally assumed to act additively in the receptor so that either their concentrations or doses (chemicals with similar action) or toxic responses (chemicals with independent joint toxic action) caused by them can be summed (e.g. Kortenkamp et al., 2009). Approaches applied in the case of concentration/dose additivity include using a toxic equivalent factor (TEF) or Hazard Index (HI). Joint toxic actions of dissimilarly acting contaminants can come up as combined toxicity that deviates from additivity, such as antagonism (combined toxicity < additive toxicity) or synergism (combined toxicity > additive toxicity). Such effects can best be studied using biological methods, i.e. bioassays or biomonitoring on site. So far, mixture effects have seldom been explicitly addressed in site-specific RAs based on exposure modeling in Finland, except in the case of contaminants considered following additivity, such as dioxins and dioxin-like PCBs (Sorvari and Assmuth, 2000; Sorvari, 2004c).

**Models and calculation tools**

In HRA associated with land contamination, risk estimates are generally determined using exposure models, complemented with specific transport models, if necessary. Exposure models require data on potential intake of different contact or exposure media involved, exposure times and frequencies, and generic properties of the receptor, such as body weight. Some of these factors (e.g. exposure

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\(^5\) Note. The United States Environmental Protection Agency uses the TRIAD concept in the context of hazardous waste site cleanup by referring to an approach that includes the following three elements: systematic project planning, dynamic work strategies, and innovative rapid sampling and analytical technologies (ITRC, 2003). The primary product of such a TRIAD approach is an accurate conceptual site model that can support decisions about exposure to contaminants, site cleanup and reuse, and long-term monitoring.
frequency) should be site-specifically determined, whereas literature (e.g. USEPA, 1989 & 1997b; ECETOC, 2001; Lijzen et al., 2001; Otte et al., 2001) and databases (e.g. Expofacts\(^6\)) can provide generic exposure parameter data.

In the USA, biokinetic models (BKM) for assessing health risks caused by soil lead have been under intensive development (e.g. Bowers et al., 1994; Bowers and Cohen, 1998; Pounds and Leggett, 1998). The results of these studies have formed the basis of the model known as IEUBK\(^7\) (USEPA, 2009a), which is applied instead of the dose versus safe intake methodology generally applied in European countries. Besides lead, the biokinetic models developed for use in HRA associated with soil contamination cover at least the following contaminants: arsenic and chromium (Lord-Hoyle et al., 2008); nickel and copper (Vasiluk and Hale, 2008); and dioxins (Kerger et al., 2007).

Nowadays it is common practice to use so-called multimedia software tools, particularly in site-specific HRA. Such tools include all models (i.e. algorithms) for assessing contaminant distribution, transport pathways, and human exposure routes in varying levels of detail. QSAR models are usually the simplest form of models used to assess contaminant distribution and uptake by biota. Sometimes it is necessary to use specific software tools to assess particular transport pathways (e.g. contaminant leaching to or transport in groundwater), or generate more detailed data, for instance, on degradation and chemical speciation of contaminants in order to increase the reliability of risk assessments.

Many European countries have developed their own multimedia software tools for assessing health risks caused by land contamination (Ferguson et al. 1998; Quercia & Mariotti, 1998; Swartjes, 2002; Bardos et al., 2003; Poletti et al., 2004; Walden, 2005b). In the USA, the Environmental Protection Agency provides an array of different tools for conducting HRA and assessing contaminant transport\(^8\). In Finland, a multimedia calculation tool was developed for assessment of health risks caused by petroleum hydrocarbons (Öljyalan palvelukeskus Oy, 2003). When other contaminants are involved, quantitative site-specific HRAs are based on the use of software tools developed abroad (Sorvari and Assmuth, 2000; Sorvari, 2004c). Several studies have shown that although the basis of multimedia HRA tools is principally the same, different tools often result in variable risk estimates even when equivalent input data are used (Rossi, 1999 & 2002; Swartjes, 2002; Poletti et al., 2004; Walden, 2005b; Chen and Ma, 2006). These disparities mainly arise from differences in the single algorithms for calculating contaminant transport and default values used in exposure assessment. Poletti et al. (2004) therefore stresses the importance of understanding the input parameters in terms of what role they play in the calculation.

In ERA, the ‘wildlife protection approach’ adopted in the USA has led to the development of several models for assessing the uptake and exposure of biota to contaminants in soil; some of these models are presented in Paper V. Judging by the literature, in Europe these uptake and exposure models are rarely applied in site-specific ERA, whereas in the USA, their use is common practice. Overall, the number of quantitative calculation tools for assessment of the ecological risk in terrestrial ecosystems is much inferior to that of HRA, and the existing tools also have some limitations which diminish their usefulness in ERA (Lu et al., 2003).

Different organizations have issued varying safe daily intake values to characterize human health risks. The variation in ecological benchmarks is even wider. In Europe, many countries base their ecological soil quality guidelines, such as soil screening values (SSV), on a predefined protection level (usually 95% and 50% of species) and the corresponding concentration determined from a species sensitivity distribution (SSD) curve that compiles the toxicity data of different species,

\(^6\) Freely available at: http://cem.jrc.it/expofacts/
\(^7\) IEUBK stands for “Integrated Exposure and Uptake Biokinetic Model (for Lead)”. USEPA also provides a free software tool that is based on the model
\(^8\) Freely available at: http://www.epa.gov/risk/guidance.htm
usually covering also adverse effects on soil processes (Carlon, 2007). The methodology for deriving ecological benchmarks used in screening-level ERA, i.e. ecological soil screening levels (Eco-SSL) applied in the USA, has been quite different, since it is based on toxicity to specific species (plants, invertebrates, birds, and mammals) and assessment of the exposure of wildlife followed by a back-calculation of safe concentrations when the risks are at an acceptable level (i.e. when the exposure corresponds to the highest dose considered to be safe, e.g. a NOEL value). It is evident that these differences in methodology result in quite different benchmarks.

According to Provoost (2006 & 2008), the variation in SSVs in Europe mainly arises from the different model algorithms, default parameter values, and selected human toxicological and ecotoxicological criteria. Other factors causing variation can include differences in endpoints, assumptions on toxicity mechanisms, methodology (e.g. threshold/non-threshold approach) applied in derivation, assessment factors used in extrapolating data, inclusion/exclusion of sensitive receptors/species, and verification by epidemiological studies. Establishing a European toxicity database was in fact recognized as one of the main development needs in the context of ERA (Bardos, 2005; JRC, 2005). Derivation of SSVs and other benchmarks to be used in decision-making also involves policy aspects related to parameters, receptors and protection level. It was concluded that due to the differences in country-specific parameters and policy elements involved, SSVs will never be uniform throughout Europe. In the case of ecological benchmarks, also the receptors they consider often differ from those dominating in a particular country. Thus, any benchmarks developed abroad are not directly applicable to Finnish conditions. It is also evident that not enough attention has been paid to considering consequential uncertainties in the site-specific RAs conducted in Finland (Sorvari and Assmuth, 1999 & 2000; Sorvari, 2004c).

Since around the mid-1990s, the use of statistical tools to conduct uncertainty analysis as a part of risk assessment of contaminated sites has been emphasized, particularly in the USA (USEPA, 1997c & 2001). Monte Carlo simulation has been the most commonly applied technique, but recently the Bayesian (subjective) approach has been increasingly adopted. Also in Finland, the use of statistical tools applying Monte Carlo simulation in site-specific HRA has become more common during the last few years (Sorvari, unpublished survey). Geographic Information Systems (GIS), i.e. mapping techniques, comprise another group of methods that are increasingly applied in monitoring and illustrating, for example the dimensions of contamination or the distribution of receptors or habitats, and furthermore, integrated with RA and overall management of contaminated sites. In Finland however, the use of GIS techniques has so far been mainly limited to research projects.

1.4.3 Decision support tools involving multiple factors

Recognition of the need to consider the multi-dimensionality and multi-objectivity and overall sustainability of practical CLM in decision-making generated vast development of DSTs based on different techniques (e.g., Bardos et al., 2003; Linkov et al., 2004). Most of these techniques fall into the category of Multi-Criteria Decision Analysis/Aid (MCDA)\textsuperscript{9}. The fundamental idea in MCDA is to systematically combine the different contributing factors (decision criteria) involved in decision-making; these often have different dimensions and units. MCDA methods allow the decision-maker to account for the importance of the contributing factors in his/her decision-making by assigning weights, i.e. numerical multipliers, which imply their importance in relation to each other; by rating each factor/objective against other factors/objectives; or by ranking the decision alternatives. Numerical data having different units can either be normalized or the measures can be unified. Monetization is the major approach in the latter case, and it forms the basis of life

\textsuperscript{9} The terminology in literature varies, and at least the following terms are used as synonyms of MCDA; multi-attribute decision analysis (MADA), multiple criteria/attribute decision analysis; multi-criteria decision-making (MCDM) and multi-criteria analysis (MCA)
cycle cost analysis and cost-benefit analysis (CBA). In MCDA tools, particularly social aspects of CLM are generally considered through monetization (e.g. Cox and Crout, 2003; Marcomini et al., 2009). At the same time, not being forced to monetize all factors is actually considered a major advantage and strength of MCDA methods (Bardos et al., 2002).

Many studies that focused on comparing different RM options relied on methods used in life cycle analysis (LCA) (Bender et al., 1998; Page et al., 1999; Hiester et al., 2003; Schrenk and Barczewski, 2003; Shakweer & Nathanail, 2003; Blanc et al., 2004; Godin et al., 2004; Toffoletto et al., 2005; Bayer and Finkel, 2006; Cadotte et al., 2007; Lesage et al., 2007). A complete LCA covers environmental consequences, i.e. effects, ‘from cradle to grave’, and it therefore extends across the whole lifetime of RM actions. These effects can be measured in different units and integrated using characterization factors for impact categories or, for example, described as the magnitude of the carbon footprint (Praamstra, 2009). In practice, due to a lack of data it is often impossible to cover all sources and dimensions of environmental effects of CSM in LCA, and some study boundaries need to be set. Such boundaries include the spatiotemporal scales, processes, and environmental impacts to be covered (e.g. whether off-site tertiary impacts are considered). The variations in system boundaries and impact categories result in different outcomes in a single CLM case (e.g. Anderson, 2003; Suèr et al., 2004).

Several CLM studies have also applied CBA, cost-effectiveness analysis (CEA), life cycle cost analysis, or risk-cost-benefit analysis in considering the overall benefits versus the monetary inputs and outputs involved in alternative remediation methods (James et al., 1996; Day et al., 1997; Wolka, 1997; Katsumata and Kastenberg, 1998; Toland et al., 1998; Hamilton and Viscusi, 1999; Khadam and Kaluarachchi, 2003; Bage et al., 2004; Linkov et al., 2004; Harbottle et al., 2006; Chen and Ma, 2007; Rosen, 2008). The benefits covered in CLM studies vary and include public welfare measured as increased land value, net effects on market goods and services, and net effects on health and ecosystem goods and services; and number of human cancer cases avoided due to remediation, among other things.

MCDA tools used in CLM usually include elements that are based on different types of DSTs, such as CBA, LCA, and risk assessment (Bardos et al., 2003; Sullivan et al., 2007). MCDA tools have been developed for a single expert’s decision-making process, but also for decision-making involving multiple stakeholders such as risk managers and laymen (group decision-making tools). Until recently, MCDA methods in the CLM context have been adopted much less in group decisions than by single decision-makers (Kiker et al., 2005).

A multi-criteria decision-making exercise associated with RM of a contaminated site generally aims to choose one of a number of RM or study method (e.g. sampling) alternatives, based on how well the alternatives rate against the set of decision criteria. Decision criteria are the major consequences related to each RM alternative, and they are the factors that ultimately drive the decision-making, such as costs and environmental benefits, the optimal RM option being the eventual objective. Here ‘optimal’ can refer to, for instance, cost-efficiency, eco-efficiency, overall sustainability, or preference when all stakeholder views are considered. Each criterion can comprise sub-criteria, generally known as attributes, such as reduced health risks and improved groundwater quality, as components of risk reduction and environmental benefits. The factors (criteria, attributes, or sub-attributes) at the lowest level must be measurable. A value tree describes the mutual hierarchy of the factors involved. Structuring of a value tree is a crucial work stage, since differences in the hierarchical structure can significantly affect the results of the analysis (e.g. Pöyhönen and Hämäläinen, 1998).

Various methods are available for constructing a decision problem, conducting weighting, and processing the results in accordance with MCDA. Different approaches can result in different

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10 Cost-effectiveness analysis (CEA) is a simplified version of CBA, differing from it in that instead of monetizing, the benefits are scored individually (Environment Agency, 1999).
indications of the best alternative or even opposite rankings of the alternatives (e.g. Triantaphyllou and Baig, 2005). Unfortunately, selection of an appropriate MCDA procedure for solving a particular decision problem is not unambiguous, and only some generic guidelines have been issued for this purpose. Compensation degree is one of the key factors (e.g. Guitouni and Martel, 1998). Compensatory methods assume that high performance of one decision criterion/attribute involved in decision-making can at least partially compensate for low performance of another criterion/attribute that meets any initial minimum performance requirements. Whereas non-compensatory methods assume that such tradeoffs are not accepted. The type and amount of data as well as the outcomes also vary in the different MCDA methods: while some rank the alternatives others identify the optimal alternative, provide an incomplete ranking, or differentiate between acceptable and unacceptable alternatives (e.g. Kiker et al., 2005). The techniques used in the context of CLM for selecting RM measures at least include Multiattribute Utility/Value Theory, i.e. MAUT/MAVT (Ralston, 1996; Timmerman et al., 1996; Beinat and van Drunen, 1997; Grelk et al., 1998; Nijboer et al., 1998; Accorsi et al., 1999; Bonano et al., 2000; Apostolakis, 2001; Parnell et al., 2001; Arvai and Gregory, 2003); Analytical Hierarchy Procedure, AHP (Accorsi et al., 1999; Bonano et al., 2000; Apostolakis, 2001; Bezama et al., 2007; Carlon et al., 2007); Stochastic Multicriteria Acceptability Analysis (SMAA) and Simple Multi-Attribute Rating Technique (SMART) (Wakeman, 2003; Bezama et al., 2007); Ideal Point Analysis (Salt and Dunsmore, 2000); Weighted Summation (Balasubramaniam et al., 2007); and outranking methods ELECTRE, ELimination Et Choix Traduisant la REalite’ (Balasubramaniam et al., 2007) and PROMETHEE, Preference Ranking Organization Method for Enrichment Evaluations (Vranes et al., 2001; Khelifi et al., 2006). Of these, MAUT/MAVT seems to be the most common, and although AHP is widely applied in other disciplines, the literature survey showed that its use in CLM has so far been limited. The study by Bello-Dambatta et al. (2009) verifies this conclusion.

MAVT/MAUT and AHP are both compensatory methods. The goal of MAUT/MAVT is to find a simple expression for the net benefits of a decision (e.g. Linkov et al., 2006). It uses utility or value functions, transforms diverse criteria into one common scale of utility/value, and aims at maximizing the latter. Similarly to MAUT/MAVT, AHP aggregates various facets of the decision problem using a single optimization function known as the objective function, the goal being selection of the alternative that results in the greatest value of the objective. AHP uses pair-wise comparisons of decision criteria instead of utility or weighting functions involved in the MAUT/MAVT method. The strengths of MAUT/MAVT include transparency and ease of comparing RM alternatives, whereas the costs of conducting rigorous preference elicitations, which accurately reflect stakeholders’ preferences (in the case of group decision-making), is a major weakness (Linkov et al., 2006). In contrast, in AHP, weighting is easy to implement but the results do not necessarily reflect stakeholders’ true preferences, a problem that was proven to originate from the rather arbitrary evaluation scale of the weights (e.g. Pöyhönen and Hämäläinen, 2001). To overcome this problem, Salo and Hämäläinen (1997) developed an AHP method that uses balanced evaluation scales. Linkov et al. (2006) further state that the mathematical procedures in AHP have resulted in illogical results and that the rankings are sometimes non-transitive.

In MAUT/MAVT, accommodating tradeoffs means that all criteria/attributes must be measured in comparable units or alternatively, that non-commensurable data describing the performance of different alternatives in terms of each criterion and attribute are normalized to a uniform measure. It is also noteworthy that the criteria and (sub-)attributes must be independent of each other. In selecting the weighting method, it needs to be acknowledged that direct weighting can be challenging if a vast number of criteria, attributes, or sub-attributes are involved. In such a case pair-wise

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11 A detailed description of the different MC(D)A techniques is beyond the scope of this research. Information on the alternative techniques can be found in various data sources, e.g. Seppälä et al. (2001).
Several of the existing MCDA methods used in CLM were developed for a specific purpose, e.g. to characterize a contaminated site, optimize and design site studies (sampling), plan monitoring actions, or optimize a particular remediation process. The latter category includes DSTs developed to determine the feasibility of phytoremediation (e.g. Japenga and Römkens, 2000; Robinson et al., 2003; Lewandowski et al., 2006; Porter et al., 2006; ETH, 2009), surfactant-enhanced remediation of soil (Huang et al., 2003), and pump-and-treat remediation of groundwater (Hoffman, 1993; Rifai et al., 1994). Several studies also developed DSTs to compare alternative groundwater treatment techniques and other options for groundwater RM, including monitoring (Aziz et al., 2003; Teutsch and Finkel, 2003; Ling et al., 2004; Huang and Wu, 2005; Khelifi et al., 2006; Huang et al., 2007; Nasiri et al., 2007). In addition, various software products not fixed to a certain remediation technique(s) or a particular environmental compartment are also available; these are summarized in Table 5 (Section 4.8.4). MCDA-based DSTs have also been integrated with GIS techniques (e.g. Salt and Dunsmore, 2000; Carlon et al., 2007). To conclude, MCDA methods have been applied in several CLM studies abroad. Only two cases were documented in Finland (Hokkanen et al., 2000; Lahdelma et al., 2001), though. In addition, Paper I and III present Finnish case studies that incorporated the MCDA technique into ERA and HRA.

2 Research aims, methodology, and boundaries

This research applied site-specific risk assessment and multi-criteria (decision) analysis to determine risks and optimal risk management actions at some contaminated sites in Finland. The separate studies included in the research primarily aimed to define the magnitude of risks at the study sites, but also to demonstrate the use of different RA methods and tools. Evaluating the suitability of these methods and tools and identifying issues relevant to conducting site-specific RA in Finland were essential elements of the research. In order to widen decision-making on CLM beyond the risk aspect, i.e. to cover other relevant factors, a multi-criteria DST was developed and tested with a couple of contaminated sites typical of Finland.

The specific research questions were:

- Does contamination at the sites examined cause significant health and/or ecological risks warranting some RM actions or more detailed site studies and/or risk assessment?
- Are the studied risk assessment methods and software tools suitable and useful for assessing risks at equivalent Finnish contaminated sites and what are their major differences and limitations?
- Could simple, i.e. screening-level, risk assessment methods provide adequate information for decision-making concerning contaminated sites?
- Can the developed multi-criteria DST support selection of RM methods? Does this tool need further development and what are its limitations?
- a) Is the TRIAD procedure applicable to assessment of ecological risk at a typical Finnish contaminated site and b) does incorporation of a statistical calculation tool and MCA provide a feasible and useful method for comprehensively assessing the different uncertainties involved in TRIAD-based ERA?

The research was carried out by conducting risk assessments of different types of contaminated sites typical of Finland, using various methods. The work concentrated on quantitative RA using alternative models and software tools for HRA, but qualitative rating, biomonitoring, bioassays,
and ecological studies were also applied. Concentration versus benchmark comparison generally used in the identification of risks and in baseline risk assessment was included, being the first stage in tiered RA and the most common RA approach in Finland, and thus a very important instrument supporting decision-making in practice. In the quantitative HRA tools, the focus was on examining

- the range of some key chemical-specific and site-specific parameters involved in calculations and the contribution of their variation to the uncertainty of risk estimates;
- their suitability for assessing the risks caused by contaminants present at the study sites;
- their suitability for assessing risks in Finnish conditions;
- the variability of reference values and benchmarks used in determining risk estimates and their effect on the results of RA.

The research did not include a detailed analysis of the algorithms and exposure parameters included in the studied HRA tools. Poletti et al. (2004) and Swartjes (2002) conducted such investigations for some European DSTs used to assess human health risks caused by land contamination. The results from these studies and other relevant comparison studies (Rossi, 1999; Butler and Petts, 2000 & 2002; Rikken et al., 2001; Rosenbaum et al., 2002; Chang et al., 2004; Walden, 2005b; Rosenbaum, 2006) were reviewed and utilized in this research. Moreover, additional methods used in the case studies, i.e. bioassays, ecological studies, and biomonitoring, as well as laboratory scale methods for examining contaminant transport or availability, were merely considered as means of producing information for risk assessments, and thus they were not evaluated in detail per se. Overall, the focus was on contact/exposure medium-dose-response assessment and the tools and methods that were used in the site-specific RAs (see Section 3.1.1, Table 1), as well as on the exposure routes and transport pathways identified as most important at the study sites. For comparison, some additional multimedia tools were also examined, however. All the materials, methods, and tools involved are described in more detail in the following section (3).

3 Materials and methods

3.1 Decision support tools

3.1.1 Risk assessment procedures and methods

The site-specific risk assessments conducted in this research followed a tiered approach (Fig 3).

Different quantitative tools and a qualitative rating system were tested in site-specific risk assessments of specific contaminated areas (i.e. study sites). The uncertainties of the site-specific RAs were assessed using a qualitative scale (rating based on own judgement) or a statistical software tool.

The main criteria in the selection of the RA methods included the availability and amount of data needed. Freely available methods and tools (e.g. via the Internet) were prioritized due to a lack of resources, but also because of their availability per se, i.e. since such tools/methods are also readily available to all risk assessors in Finland, and hence, could be in extensive use in the future (if not already at present). In addition, the research included tools that were readily available in the Finnish Environment Institute where the case studies were conducted. Additional factors considered in the selection included the following:

- characteristics of the study site: contaminants, size of the area, land use;
- type of tool, i.e. whether it is meant for a screening-level or more detailed assessment;
- type of input data needed (depends on the type of tool);
- type of outputs; and
- ease of use (Windows-based software was prioritized).
The tools used in HRAs therefore included the freely available SSL calculator and Risc-Human multimedia software. The latter was used to derive the current health risk-based Finnish soil quality benchmarks (Reinikainen, 2007) and it has also been applied in several previous site-specific HRAs in Finland. Additional methods included generic distribution and transport, uptake, and exposure algorithms, which were input into Excel to perform the calculations (= manual calculations). Specific hydrological transport and geochemical distribution and speciation models were not investigated, as this would be a study of its own. In fact, in practically all the study sites the available quantitative data were inadequate for using such detailed models. It is also worth mentioning, that this research focused more on examining the differences in multimedia model variation in terms of exposure parameters, mainly because the different level of detail of the tools meant that the route- and pathway-specific models were not directly comparable.

The choice of quantitative tools for conducting ERAs was limited. Moreover, in most of the projects where the case studies were conducted, insufficient resources had been allocated to ERA. This meant that also the data needed by ERA was in most cases inadequate to conduct other than tier 0 and tier 1 risk assessments, with one exception, however. The study of a former landfill site was a project funded by the EU-LIFE program, where it was possible to demonstrate the use of multiple methods for conducting ERA, i.e. chemical and biological studies. The participating researchers selected the latter methods and conducted the corresponding studies included in the project.

12 The division of the RA procedure into separate tiers varies: in some frameworks, tier 0 is considered the work step where a conceptual model is built, while tier 1 involves comparisons of environmental concentrations against suitable benchmarks. In this research, the latter phase is considered to belong to tier 0 and tier 1 covers site-specific calculations.

13 Covers, for example, models for assessing human exposure to contaminants (route-specific models), such as food intake and inhalation, and models for determining the transport of contaminants (pathway-specific models), for example from soil to groundwater and from soil to plant.
Several software tools are available for conducting uncertainty analysis. The differences in these tools are mainly associated with practical issues (e.g. how the input data are entered and what are the outputs), technical properties (e.g. whether they are stand-alone tools or can be linked with Excel), the number of available statistical distributions, and the types of sampling techniques involved. This research applied CrystalBall™ software, since it was readily available in the Finnish Environment Institute. CrystalBall™ is an add-in software that runs in Excel.

Table 1 presents a summary of the RA methods and tools and where they were applied in this research.

Table 1. Risk assessment methods and tools used in this research. UA = uncertainty analysis, GW = groundwater

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<tr>
<th>Tool or method</th>
<th>RA type and level</th>
<th>Paper</th>
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<tbody>
<tr>
<td>Benchmarks*</td>
<td>ERA, Tier 0</td>
<td>I, III</td>
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<td></td>
<td>HRA related to GW contamination</td>
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<td>QSAR models (manual calculations)</td>
<td>RA of GW contamination</td>
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<td>HRA and ERA (plant uptake), Tier 1</td>
<td>V</td>
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<td>SSL calculator† (USEPA)</td>
<td>HRA, Tier 1</td>
<td>II</td>
</tr>
<tr>
<td>Risc Human 3.1. software (Van Hall Institute, 2000)</td>
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<td>II*, III</td>
</tr>
<tr>
<td>Uptake models*</td>
<td>ERA, Tier 1</td>
<td>V</td>
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<td>Exposure models*</td>
<td>ERA, Tier 1</td>
<td>V</td>
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<td></td>
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<td>Biokinetic model (manual calculations)</td>
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<td></td>
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<td>CrystalBall™ software</td>
<td>ERA, UA in Tier 2</td>
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<td>VI</td>
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<tr>
<td>Qualitative rating</td>
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<td></td>
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<td>Solubility tests</td>
<td>RA of GW and future soil contamination</td>
<td>V</td>
</tr>
<tr>
<td></td>
<td>ERA, Tier 1 (assessment of potential availability of contaminants)</td>
<td>I, V</td>
</tr>
</tbody>
</table>

*various benchmarks compiled from the literature
†SSL = Soil Screening Level; the tool is freely available at: http://rais.ornl.gov/calc_start.shtml
‡models for estimating contaminant uptake by earthworms and small mammals (terrestrial), manual calculations
§generic models for humans and small mammals (terrestrial), manual calculations
ʰin this case study the quantitative uncertainty analysis did not involve the use of statistical tools
⁻includes analyses of contaminant concentrations in plants at the sites presented in Papers I and V, but the results are presented in detail elsewhere (Naumanen et al., 2002; Sorvari, 2006)
⁻⁻the results from the rating formed the starting point for a more detailed RA presented in Paper V, but they – as well as the rating system – are documented in detail elsewhere (Naumanen et al., 2002).
3.1.2 Multi-Criteria Analysis (MCA) methods

The following methods and tools were applied in the studies focused on the application of decision analysis techniques:

- TRIAD combined with MC(D)A, used in ecological risk assessment; and
- PIRTU, an Excel-based calculation tool used in the identification of the preferable risk management alternatives for two types of contaminated sites.

In TRIAD (Paper I), the results from different methods for assessing ecological risks were combined using calculation rules published in the literature (e.g. Jensen and Mesman, 2006). The methods included chemical analyses, various biotests, and studies of soil invertebrates. The individual bioassays were also rated according to their performance in terms of specific assessment criteria, which were ranked using weighting.

The need for an MCDA tool for CLM in Finland was identified in the studies described in Paper IV. Therefore, a freely available DST suitable for Finnish conditions was developed using the Dutch REC system (Beinat and van Drunen, 1997) as a starting point. The principles of this DST, known as PIRTU \(^{14}\), are briefly presented in Paper III and in Sorvari et al. (2005 & 2006) and Sorvari (2007). The PIRTU tool (Fig. 4) can be used to site-specifically identify the most feasible RM techniques based on different overall objectives, such as eco-efficiency, cost-efficiency, or stakeholders’ preferences, and hence it also serves as a group decision-making tool. The tool applies the MAVT technique and generates numerical scores that indicate the overall merits of alternative RM approaches. The decision criteria of PIRTU include risk reduction, environmental effects, costs, and other factors (e.g. socio-cultural effects, maintaining a positive public image). Determination of numeric data for the input values needed in the calculations is based on the use of risk assessment, life cycle analysis, cost estimation, and expert judgement combined with rating. Testing and demonstrating the tool at sites representative of Finnish contaminated sites (Paper III) was crucial in order to find out its possible further development needs, feasibility, and usefulness in an actual decision-making process involving different stakeholders.

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**Figure 4.** Decision criteria included in the PIRTU calculation tool (modified from the original figure by Riina Antikainen).

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\(^{14}\) Freely available at: http://www.ymparisto.fi/syke/pirre
The weighting methods applied in the MCA procedures included direct weighting in accordance with SMART (Edwards, 1977) and pair-wise weighting as per the AHP technique (Saaty, 1980), using the HIPRE\textsuperscript{15} software of the Helsinki University of Technology, Laboratory for Systems Analysis (Paper III) and a trial version of RightChoiceDSS 2.1.0.21\textsuperscript{16} software by Tier3, Inc (Paper I). The latter software was also used to generate the final performance scores for each ERA method on the basis of weights and performance values produced using direct ranking of the assessment criteria. All other calculations were run in Excel, including the uncertainty analysis using CrystalBall\textsuperscript{TM} software (Paper I).

It is noteworthy that the MCA procedures used in the identification of the most preferred (or eco-efficient) RM alternative (Paper III) and in the site-specific ERA following the TRIAD methodology (Paper I) had different starting points. In the former, the aim was to consider different stakeholders’ viewpoints (group decision-making) and facilitate communication between stakeholders, thereby increasing their understanding of the effects of different preferences on the final result, whereas in the latter study the purpose was to objectively rank the study methods (rating and weighting conducted by single experts) in order to increase the reliability of risk estimates.

3.2 Study sites

The study sites included in this research were contaminated by different chemicals (Table 2). These represented, besides the major types of contaminated sites in Finland, also difficult sites (from the viewpoint of risk management) and sites that had hardly been systematically studied before. Although both HRA and ERA were conducted at all study sites, some results have been excluded from this research and are presented elsewhere (Hellman \textit{et al.}, 2002 & 2003; Schultz \textit{et al.}, 2006; Sorvari, 2006).

<table>
<thead>
<tr>
<th>Type of contaminated site</th>
<th>Studies</th>
<th>Additional information</th>
<th>Paper</th>
</tr>
</thead>
<tbody>
<tr>
<td>Industrial landfill</td>
<td>Site-specific ERA using the TRIAD approach, statistical uncertainty analysis, and the MCDA procedure</td>
<td>Actual former landfill, focus on soil contamination</td>
<td>I</td>
</tr>
<tr>
<td>Forest nursery</td>
<td>Site-specific RAs (HRA, ERA), calculation of SSLs</td>
<td>Two actual sites in different locations; all environmental compartments considered</td>
<td>II</td>
</tr>
<tr>
<td>Former gasoline station</td>
<td>Determination of the most preferable RM option using the PIRTU tool, including HRA, ERA, LCA, cost analysis; assessment of social effects</td>
<td>Fictitious site created using data from actual sites</td>
<td>III</td>
</tr>
<tr>
<td>Shooting range</td>
<td>Site-specific HRA, ERA, and assessment of groundwater pollution risk</td>
<td>Three actual sites in different locations; all environmental compartments considered</td>
<td>V</td>
</tr>
<tr>
<td></td>
<td>Determination of the most preferable RM option using the PIRTU tool, including HRA, ERA, LCA, cost analysis; assessment of social effects</td>
<td>Fictitious site created using data from actual sites</td>
<td>III</td>
</tr>
<tr>
<td>Residential area contaminated by PCBs</td>
<td>Site-specific HRA based on generic exposure models, statistical uncertainty analysis, and biomonitoring</td>
<td>Study around 11 concrete buildings where PCB sealants were previously used</td>
<td>VI</td>
</tr>
</tbody>
</table>

\textsuperscript{15} Available at: \url{http://www.hipre.hut.fi}

\textsuperscript{16} Available at several sources, e.g. \url{http://www.bestshareware.net/download/rightchoicedss-professional.htm}
The research documented in Papers I, II, V, and VI dealt with actual sites, whereas the study described in Paper III used fictitious data created on the basis of similar real cases. The reason for this was the different objectives of the latter study. While the research involving site-specific RAs was focused on demonstrating the use of a tiered RA procedure and different RA methods and producing information for the selection of actual RM measures at those particular sites, the study described in Paper III aimed to test the multi-criteria DST (PIRTU) developed for Finnish conditions. This study required quantitative data on the costs, environmental effects, and other consequences (e.g. social effects and effects on the attractiveness of the site) of alternative RM techniques. Such data were not readily available from the actual cases and therefore, they needed to be compiled from various sources.

3.3 Methods used to study the state of the art in CLM

The research also included a study on the factors that contribute to present RM practices in Finland, possible problems involved, e.g. in the practices used to assess risks, environmental effects, and costs when selecting RM measures, and a brief review of DSSs in use abroad (Paper IV). This study was based on analyzing the data generated using the following methods:

- Metaplan group working method;
- thematic interviews;
- questionnaire study;
- literature and Internet surveys; and
- survey of previously documented Finnish remediation projects.

Metaplan, i.e. a card technique, is a working method in which brainstorming by a group of participants is facilitated by a moderator, and each participant’s independent ideas are processed and organized according to relevant categories and further discussed within the group. The implementation of this study, the studied data sources, and the contents and recipients of the thematic interviews and questionnaire study are detailed in Paper IV.

4 Results and discussion

4.1 Risks and risk management needs at the study sites

This section briefly presents the results from the site-specific risk assessments of the selected study sites and the identified risk management needs.

*Former industrial landfill site*

The TRIAD-based ERA conducted at a former industrial landfill site to determine the risks to terrestrial biota produced high risk estimates (Paper I). The results from separate ERA studies were, however, partly controversial and not clearly supported by field observations. Quite expectably, the bioassays and studies of soil invertebrates showed that assessment based solely on chemical studies that fail to consider several aspects, such as limited bioavailability of contaminants, and adaptation or avoidance of receptors, overestimated risks. Some practical problems also appeared in the biological studies, a lack of suitable reference soil being the most important. The hormesis effect caused additional uncertainty in some ecotoxicity tests. MCA incorporated into the TRIAD procedure proved to be a useful way to account for the reliability of the ERA methods not covered in the statistical analysis based on Monte Carlo simulation. In this study, adopting MCA only resulted
in minor differences in the final integrated risk estimates, however. Risk to aquatic ecosystems was not part of the study, but was identified as a potential issue. Removal of soil in the identified hot spot containing petroleum hydrocarbons was recommended in order to limit further contaminant transport to groundwater and the adjacent sea. Otherwise, no risk management measures were suggested to limit ecological risks before the consequences of alternative RM actions have been systematically evaluated. On the other hand, HRA (not documented in this research) showed that land use restrictions are needed in order to limit human exposure to metals (Sorvari, 2006).

Forest nurseries

Studies at two forest nursery sites (Paper II) revealed that contaminated groundwater at Site 2 poses an immediate risk to human health if used as drinking water. HRA showed additional health risks associated with soil contamination if the sites are taken into residential use in the future. These risks can be efficiently eliminated by replacing the topsoil of the spatially restricted hot spots. Tier 1 HRA provided adequate information for identifying RM needs and feasible RM actions, and no further assessment was therefore needed. Due to the diversity and number of chemicals involved and the lack of toxicity data, risks to terrestrial ecosystems could not be reliably identified using only chemical data (tier 0). Nevertheless, since chemical studies showed mainly low pesticide concentrations in the assumed hot spot areas, risks to terrestrial ecosystems are expected to be minor and not to warrant more detailed studies. Chemical studies showed that some pesticides had migrated along runoffs and therefore, verification of potential transport to adjacent surface water and the resulting risks to aquatic ecosystem by means of additional site studies and control of runoffs was recommended.

Shooting ranges

Site-specific RAs at two shooting ranges showed that major risks to human health arise from potential contamination of groundwater used to supply household water (Paper V). The time span for contaminants to reach groundwater depends on the soil type and the depth of the groundwater table. The results from leaching tests and extrapolation using conservative assumptions indicated that in the case of the main contaminant, lead, it varies from a couple of years to a century. The corrosion of ammunition is the key factor from the viewpoint of future contamination of soil and other environmental compartments and consequent risks, and it was estimated to extend even to a millennium, at the maximum. Monitoring groundwater quality proved to be the minimum RM measure, and closing of groundwater abstraction was recommended. In recreational use, the main risk arose from potential soil ingestion and consumption of contaminated wild berries and mushrooms. These risks could be minimized by restricting recreational activities and thus, no active remedial actions were necessary. The tier 1 HRA provided adequate data for RM and therefore, no higher tier assessment was warranted. Tier 1 ERA showed partly significant risks to biota. The risk estimates were based on rather conservative assumptions, however, and they also included several uncertainties, which could not be quantitatively determined. Hence, verification of the magnitude and scale of risks by conducting a more detailed ERA, preferably using bioassays combined with biomonitoring, was recommended.

Residential areas contaminated by PCB

The study dealing with soil contamination around concrete buildings with PCB-containing sealants showed elevated PCB concentrations, which quickly declined with increasing distance from the buildings. On the basis of deterministic calculations using generic exposure algorithms, the
residents’ lifetime cancer risks and non-cancer risks would remain low, the average daily exposure to PCB in soil being around one tenth of the average Finnish daily dose from foodstuffs. Statistical uncertainty analysis showed that these results represented approximately the 90th percentile in the probability density curve. The results were verified by biomonitoring studies based on analysing residents’ blood serum, since these showed no statistically significant differences compared with the control group. However, according to the worst-case exposure scenario, young children could be exposed to levels equivalent to the safe daily doses due to their common hand-to-mouth behavior. No immediate RM actions to protect residents were suggested, and no further studies or more detailed HRA were warranted. Some feasible RM measures, such as replacing the sandpits and soil in playgrounds and restricting the cultivation of edible plants adjacent to buildings with PCB-containing sealants, were recommended in order to minimize human exposure.

**Identified main risk factors**

Soil ingestion and/or food consumption were the most important human exposure routes of contaminants in all the study sites (Paper II, V, VI; Sorvari, 2006). This result is in line with the Dutch studies using the CSOIL model (this model forms the basis for Risc-Human), which showed that these are the major contributors to total exposure to the non-volatile contaminants (Lijzen et al., 2001). At some study sites, contaminant concentration(s) in groundwater exceeded the corresponding quality standard(s) for domestic water, thereby indicating significant risks to human health when used as tap water. In addition, the calculations and leaching tests showed risks to groundwater contamination at the shooting ranges (Paper V). Dermal intake was the main exposure route (besides soil ingestion) in the case of PCB contamination (Paper VI), whereas inhalation contributed significantly to total exposure only in the case of some pesticides (Paper II). The type of land use (scenarios) and physicochemical properties of the contaminants mainly explain these differences.

### 4.2 Investigation of the risk assessment methods used in tier 0 and tier 1

#### 4.2.1 Benchmarks and qualitative rating of risk components

The tier 0 risk assessments included qualitative rating and comparisons of environmental concentrations of COPCs/COPECs against different concentration limits, such as ecological benchmarks based on risks to specific organisms or soil biota in general, and land-use-specific limit values protective to human health. Such benchmarks vary considerably, and consequently, the differences in risk estimates can reach several orders of magnitude (as shown in Paper II and V; also Sorvari et al., 2007). This variation has been recognized in several studies (e.g. Provoost et al., 2006 & 2008; Carlon, 2007).

A qualitative rating system was used to identify and systematically screen the key transport pathways, exposure routes, receptors, and contaminants at shooting ranges (presented in Naumanen et al., 2002, pp. 92-94). The results thereby formed the basis for building site-specific conceptual models. The results allowed the exclusion of some human exposure routes, i.e. those associated with surface water contamination and dermal absorption. Since the qualitative rating did not produce information on the magnitude of exposure, more detailed information on the risks was needed to define whether any RM actions are required and to identify alternative RM approaches (Paper V). According to Cox et al. (2005), qualitative ratings can actually result in errors, which appear as reversed ranking (assigning higher qualitative ratings to quantitatively smaller risks) and uninformative ratings, i.e. equivalent ratings for risks that differ by several orders of magnitude. This research verified the conclusion that qualitative ratings would not provide sufficient infor-
mation to discriminate accurately between risks of differing magnitude. Cox et al. consequently recommended the use of simple quantitative models instead of such ratings.

4.2.2 QSAR and empirical distribution, transport, and uptake models

QSAR models were applied in the assessment of contaminant transport from soil to groundwater (Paper II and V) and plant uptake. In addition, purely empirical models, presented in the literature, were used to assess plant uptake of metals. The detailed results of these plant uptake studies are documented elsewhere (Naumanen et al., 2002; Jaakkonen and Sorvari, 2006; Sorvari, 2006) and not reported here. In addition, validated empirical uptake models based on uptake factors and simple regression functions were used in ERA to assess uptake of metals by earthworms (Paper V; also Sorvari et al., 2007).

High uncertainty is generally inherent in the estimation of leaching of contaminants from soil to groundwater when QSAR models based on partition coefficients of contaminants between soil solids and soil water are used in the assessment. These coefficients are chemical-specific, and soil properties also affect their magnitude. In practice, the variability of the soil solids-water partition coefficient (Kd) value and consequently, the estimated concentration in soil water and groundwater, can extend to several orders of magnitude (Paper II). The Kd values of metals also change when pH changes, which further reduces the usability of literature values. Considerable variability of the hydraulic conductivity of soil (k), which is used in the calculation of a dilution factor between soil water and groundwater, causes even more uncertainty in the estimation of leaching to groundwater (Paper II and V). The lack of site-specific data is therefore an issue when applying QSAR-based distribution models that rely on k and Kd values. In the case of organic contaminants, equivalent distribution models use organic carbon-water coefficients (Koc). Koc values can be found in various databases and literature, but as shown in Paper II, they vary in different sources. The problem of the variation in physicochemical parameters (and its manifestation in risk estimates) has in fact been recognized in several studies. Marino (2006) studied eight standard sources of chemical data and noticed that particularly octanol-water partition coefficient (Kow), vapor pressure, solubility, and Henry’s law constant, i.e. the values considered to be most important parameters in evaluating the environmental fate of contaminants, exhibited considerable variability. This variation is reflected in the default values in multimedia HRA tools used in CLM (Swartjes, 2002). Linkov et al. (2005) also stated that the variation in Kow can affect the remedial targets, which can consequently result in significant financial implications for remediation. The need for verified Kow, Kd, and other physicochemical data is thus evident.

QSAR models predicting plant uptake can be either theory-based or found purely on empirical studies. Plant uptake of organic contaminants in particular, is generally assessed using QSAR models. Most models applied in HRA are based on Kow values as a predictor, but some models use molecular weight instead (e.g. Topp et al., 1986). It is worth noting that generic QSAR models are poorly applicable in assessing uptake of ionizing organic chemicals, for which specific models should be used. Some models describe contaminant accumulation in the whole plant, while others separate uptake by the roots and other plant parts. In practice, plant uptake varies among different plant species and in addition, soil properties and chemical speciation of the contaminants affect uptake (e.g. Cornelis and Bierkens, 2005). Models developed on the basis of empirical studies are therefore strictly suitable only in situations where all these variables are similar or when they can be replaced with site-specific values in the model. Moreover, climatic conditions provide boundaries to plant growth and consequently, contaminant uptake. Several studies abroad have shown that most plant uptake models, whether based on the theoretical QSAR approach or derived from empirical studies, tend to overestimate plant concentrations (e.g. Versluijs and Otte, 2001; Collins et al., 2006). In the study on shooting ranges, comparison of the measured concentrations
of lead and arsenic in plants against concentrations calculated using validated empirical uptake models showed that the latter were about three orders of magnitude higher compared with the measured concentrations, at the maximum, the difference varying depending on the plant and contaminant (Naumanen et al., 2002, p. 185). On the other hand, in the HRA included in the study on a former landfill (site described in Paper I), using the concentrations of lead, chromium, and cadmium detected in mushrooms or calculated by applying empirical uptake models resulted in almost equivalent daily dose estimates, despite the fact that the models were not specifically derived for mushrooms (Sorvari, 2006). Some Finnish studies also, showed, unexpectedly that the arsenic (As) concentrations in earthworms determined using uptake models and mean As concentrations in soil were rather congruent with the As concentrations analyzed in depurated earthworms (difference < 60%) in three different areas with varying soil As levels (Sorvari et al., 2007, Table 25). Nevertheless, further studies with higher numbers of samples are needed in order to verify these results and to study their congruence in the case of other contaminants. These results show that no definitive conclusions can be drawn on the validity of the studied empirical models in Finnish conditions, but they can at least serve as preliminary RA tools in identifying risks and determining whether more detailed site studies are warranted.

4.2.3 Exposure models

This section summarizes the methods used in exposure calculations in tier 1 risk assessments, and includes a brief comparison of alternative methods. The key input parameters involved are discussed in more detail under Section 4.3.3.

Assessment of human exposure

Generic exposure models were used to determine human health risks by manually calculating exposure estimates (Paper V and VI). These calculations were based on the equivalent algorithms built into the multimedia software tools used in HRA, with the exception that the latter include specific algorithms for determining contaminant distribution in different environmental compartments and factors that account for limitations of exposure and bioavailability.

As presented above, contaminant intake via soil ingestion was as significant route of human exposure at all the study sites (Paper II, V, and VI). This route was dominating particularly in the case of children. The amount of ingested soil is consequently a key contributor to the final risk estimates. Several studies have been conducted to determine a correct value for this parameter. The variability of the study methods has resulted in variable soil ingestion estimates, which is also to some extent reflected in the default values used in the multimedia HRA tools (see section 4.3.3.).

Major uncertainties associated with estimates of exposure via drinking water are expected to arise from the method used to determine contaminant concentration in groundwater (see section 4.2.2) - that is if measurement data are missing - and the value for drinking water intake. The uncertainties will mainly arise from the former since representative intake values can be found in Finnish food statistics.

In the case of exposure to contaminants in food, data on the consumption of food items are a major contributor in final risk estimates besides the plant uptake models (section 4.2.2). Statistics for average Finnish food consumption are available for various food items and applicable in HRA. These statistics are not exhaustive, however, and lack data on, for example mushrooms. This causes additional uncertainty in HRA if mushrooms are involved as a potential source of contaminants.

There are two different approaches to assessing dermal exposure caused by contact with soil, namely using permeability coefficients, which express the rate of skin penetration, or absorption factors, which indicate the fraction of the applied dose absorbed across the skin over a specified
period of time (USEPA, 1992). USEPA recommends using the absorption fraction-based method when assessing dermal exposure to soil or sediment, because permeability constants have been derived in aqueous solutions, and there is no evidence of their applicability in other contact media (USEPA, 1992 & 1997b). This method was therefore adopted in this research to determine dermal intake from PCB-contaminated soil (Paper VI). Both methods, whether based on permeability coefficients or absorption coefficients, include various uncertainties caused by the variability of parameter values, such as the area of skin in contact with soil, the amount of soil adsorbed on the skin surface, and skin permeability. Soil type also affects adherence to skin. As far as is known, no experimental studies on dermal absorption from soil have been conducted in Finland. Permeability coefficients are generally used in the case of dermal contact with water. Differences in the methods then arise from the different starting points, i.e. whether a steady state is assumed. USEPA recommends using the steady state approach only for inorganics, whereas for organics a nonsteady-state model should be applied. The reason for such recommendation is that the nonsteady-state approach was developed for organics which exhibit octanol-water partitioning. It is therefore not applicable to inorganic chemicals. The nonsteady-state approach has been justified by the fact that contact time (during swimming, bathing) is normally too short to reach a steady state, and uptake can also occur after the exposure event, owing to the absorption of residual chemicals trapped in the skin.

The models used to assess human exposure through inhalation use inhalation rate, exposure duration, and concentration data on contaminants in air (particles and vapors). Inhalation rate varies depending on the activity and the receptor’s characteristics, such as age and gender, resulting in variability in the final risk estimate. The problem with using a simple, generic exposure model is the lack of site-specific concentration data, since the models used in tier 1 HRA in this research do not include any algorithms for assessing contaminant transport from soil to air. Since measurement of actual concentrations using, for example, air particle collectors was not an option, air particle concentrations were derived from some Finnish measurements during remediation work at other sites with a similar soil type (Paper V). This obviously led to very conservative exposure estimates for inhalation, since the study sites were not expected to involve such rough activities in the long term. The results showed however, only a minor contribution of inhalation to total exposure, most probably owing to the land use scenario (recreation), which led to shorter exposure times (compared with residential use).

Using two alternative approaches, i.e. a biokinetic model and calculation of dose versus acceptable intake ratio, in assessing human health risks caused by lead in soil produced slightly different results (Paper V). Variability of the reference values impeded the comparison of the results, and it was thus not possible to state which of the methods produces higher risk estimates. The BKM also includes several variables whose applicability in Finnish conditions was unknown and non-verifiable due to a lack of data.

Assessment of ecological risks

Generic exposure models equivalent to those used in HRA were used to assess exposure of small mammals (shrews) to metals at shooting ranges (Paper V). Although such models can supply risk assessors with information on potential risks to biota, the results include high uncertainties particularly because information on their suitability in Finnish conditions is missing. The variability of toxicity data, e.g. NOEL values, used in risk characterization creates additional uncertainty. It is also generally known that contaminant uptake and exposure vary even among earthworm species, let alone among species at higher trophic levels, such as mammals. Since no biomonitoring (except for analyzing concentrations of COPCs in plants) was conducted as a part of the site studies, it was not possible to examine the validity of the applied models.
4.3 Comparison of the multimedia software tools

4.3.1 Generic features

Table 3 presents a comparison of the multimedia HRA tools used in the case studies involved in this research and some other multimedia tools known to be frequently used by Finnish consultants. The latter tools include CalTOX, provided by the California Department of Toxic Substances Control, and the Finnish SOILIRISK, which is based on the Risk-Based Corrective Action (RBCA) methodology developed by the American Society for Testing and Materials (ASTM). Later on, the use of the commercial RISC WorkBench software (not evaluated in this research) has become more common in Finland (Reinikainen, J., personal communication 23 March 2010). The RAIS calculator (University of Tennessee, USA) was included in the comparison due to its availability and because it covers all the relevant exposure routes. The studied multimedia tools significantly differ from each other in their level of detail. CalTOX is clearly most versatile, including a vast number of transport pathways and exposure routes, whereas the SSL calculator is most restricted. CalTOX also enables examination of the time span of contaminant transport.

One of the most important limitations of Risc-Human and the calculation tools available on the Internet (SSL, RAIS) is being a stand-alone program, which cannot be linked with any other software tools used to process and store data, such as Excel. The possibility to link these multimedia tools with statistical tools to conduct sensitivity and uncertainty analysis would also increase their usability.

As shown in Table 3, some of the multimedia HRA tools, such as CalTOX, include a rather high number of input variables and therefore, require significant amount of site-specific data. Often these data are not readily available or producing them would at least require considerable resources, and even investigating the applicability of the default values and models in Finnish conditions would necessitate an extensive study of its own. At the same time, using input data whose relevance to site conditions is unknown substantially increases the uncertainties of risk estimates and cannot be considered sensible in the higher assessment tiers. CalTOX was originally excluded from the case studies of this research primarily because of its substantial data requirement. Some previous Finnish case studies also showed that application of CalTOX in the HRAs of some dioxin-contaminated sites can result in biased results (Sorvari, 2001a; Assmuth and Sorvari, 2003; Saukkonen, 2003). These biases were mainly associated with dermal exposure, which should be an insignificant exposure pathway in the case of highly chlorinated, lipophilic dioxins dominating at sawmill sites. As pointed out by Sorvari and Assmuth (1999) and Assmuth and Sorvari (2003), such biases do not, however, necessarily rise from the unsuitability of models or faulty parameter values, but rather from difficulties and shortcomings in problem formulation and interpretation of the results.

Some calculation tools (e.g. SSL and RAIS) do not allow the use of truly site-specific input values to describe climatic conditions. The SSL and RAIS calculators are both designed for preliminary assessment of contaminated sites in the USA and therefore, some of their default values are not valid in Finland. CalTOX is also made for the same purpose as these tools, but it allows changing of all input variables. Although the results presented in Paper II suggest only a minor contribution of the variability of meteorological data to variation in SSL values and consequently, to tier 1 risk estimates, this limitation reduces the applicability of the SSL and RAIS calculators in Finland.

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18 Information based on a survey (unpublished) of the RA reports available in the Finnish Environment Institute and the documents associated with remedial decisions available in the Environmental Administration data bank (https://www.ymparisto.fi; search terms: maaperä, kunnostus, riskinarviointi) as of 31 January 2010, survey conducted by J. Sorvari
### Table 3. Comparison of some multimedia software tools used to assess human health risks at contaminated sites in Finland. The comparison is based on an examination of the tools and complemented by studying relevant literature sources (CalEPA, 1993; Rossi, 1999; 2002; Rikken et al., 2001; Swartjes, 2002; Saukkonen, 2003; Öljyalan palvelukeskus Oy, 2003; Chang et al., 2004; Poletti et al., 2004) + = included, - = not included. PRG = preliminary remediation goal

#### 3A. General characteristics

<table>
<thead>
<tr>
<th>Feature</th>
<th>RAIS Chemical calculator</th>
<th>SSL calculator</th>
<th>Risc Human 3.1&lt;sup&gt;a&lt;/sup&gt;</th>
<th>CalTOX 4.0 (beta)</th>
<th>SOILRISK 1.0&lt;sup&gt;d&lt;/sup&gt;</th>
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<tbody>
<tr>
<td><strong>Primary application</strong></td>
<td>Determination of prelimi-</td>
<td>Determination of pre-</td>
<td>Determination of average</td>
<td>Determination of site-</td>
<td>Determination of site-spe-</td>
</tr>
<tr>
<td></td>
<td>nary risk estimates (hazard</td>
<td>nary site-specific soil</td>
<td>daily doses and risk esti-</td>
<td>specific target concentrations</td>
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<tr>
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<td>screening levels (SSLs)&lt;sup&gt;b&lt;/sup&gt;</td>
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<td>for soil and their exceed-</td>
<td>for soil and their exceed-</td>
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<td></td>
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<td>ance (%)</td>
<td>ance (%)</td>
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<tr>
<td></td>
<td>tool for calculating PRGs</td>
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<td></td>
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<td></td>
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<td>3.2 incl. support for 1 yr</td>
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<td>rate report; guidance on</td>
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<td>ORNL.GOV/TOOLS]</td>
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<td>[<a href="http://www.risc-site.nl">http://www.risc-site.nl</a>]</td>
<td>some of the input values in</td>
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<td>extensive size of work sheets and</td>
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<td>dispersed data</td>
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<td>Only a limited manual</td>
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<td>In a separate report (avail-</td>
<td>Algorithms presented un-</td>
<td>In a separate manual</td>
<td></td>
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<td>pages</td>
<td>able on the Internet</td>
<td>der the built-in ‘Knowledge’</td>
<td>In a separate report (at-</td>
<td></td>
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<td>module</td>
<td>tachment), including docu-</td>
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</tr>
<tr>
<td><strong>Instructions and checking of input data</strong></td>
<td>Default values (USA); no</td>
<td>Default values (USA);</td>
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<td>Default values (Finland,</td>
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<td>well described and input</td>
<td>USA, the Netherlands)</td>
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<td>adequate control of input</td>
<td>input values not given; inadequate</td>
<td>values are checked: Dutch</td>
<td>given, including guidance on</td>
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<td>values, allows entering</td>
<td>control of input values</td>
<td>default values and ranges</td>
<td>how to define values which</td>
<td>how to define values which</td>
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<td>senseless values in some</td>
<td>(in some cases possible to</td>
<td>of acceptable input values</td>
<td>always need to be deter-</td>
<td>always need to be deter-</td>
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<td>Yes, any Excel -based software</td>
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<td></td>
<td></td>
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<td></td>
</tr>
</tbody>
</table>

<sup>a</sup> Developed by Rick et al. (2001);<br><sup>b</sup> Includes PRG calculations;<br><sup>c</sup> Includes target concentrations for soil and their exceedance (%);<br><sup>d</sup> Developed by Öljyalan palvelukeskus Oy (2003).
<table>
<thead>
<tr>
<th>Feature</th>
<th>RAIS Chemical calculator</th>
<th>SSL calculator</th>
<th>Risc Human 3.1&lt;sup&gt;a&lt;/sup&gt;</th>
<th>CalTOX 4.0 (beta)</th>
<th>SOILIRISK 1.0&lt;sup&gt;d&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Format of outputs</td>
<td>On screen</td>
<td>On screen as such or as text rows</td>
<td>Different output and report modes with different levels of detail, graphs, and tables (non-modifiable, can be copied as objects)</td>
<td>Graphs, tables (can be modified)</td>
<td>Tables only</td>
</tr>
<tr>
<td>Alternative algorithms</td>
<td>No alternatives</td>
<td>Two alternative ways to determine transport of volatiles and transport to groundwater</td>
<td>Alternative models to calculate transport of volatiles (CSOIL and VOLASOIL)</td>
<td>No alternatives</td>
<td>No alternatives</td>
</tr>
<tr>
<td>Identified specific limitations and restrictions&lt;sup&gt;c&lt;/sup&gt;</td>
<td>Does not cover all transport pathways (see Table 3B). Some input values cannot be adjusted for Finnish conditions.</td>
<td>Does not cover all exposure routes (see Table 3B). Some input values cannot be adjusted for Finnish conditions. The calculation methods for assessing transport of organic contaminants produce overly conservative risk estimates.</td>
<td>Not suitable for assessing soil-to-groundwater transport. Estimation of plant uptake unsuitable for very lipophilic contaminants.</td>
<td>The total number of site-specific input parameters is very high (altogether 170 values). Produces illogical results in the case of very lipophilic chemicals (dioxins).</td>
<td>Only for HRA at sites contaminated by petroleum hydrocarbons (PHC). Difficulty in tracking risks related to separate exposure routes.</td>
</tr>
<tr>
<td>Consideration of different soil layers</td>
<td>No; soil and concentration of contaminants assumed to be homogenously distributed in the vertical direction</td>
<td>No; soil and concentration of contaminants assumed to be homogenously distributed in the vertical direction</td>
<td>Yes; possible to consider different soil depths separately; this only affects the calculation of exposure through inhalation</td>
<td>Yes; surface soil, root zone, and vadose zone studied separately</td>
<td>Yes; surface and bottom soil separated (exp.) but assumed homogenous; vadose, saturated zone, and capillary fringe separated (transport)</td>
</tr>
<tr>
<td>Choice of contaminants and source of physicochemical &amp; effect data</td>
<td>Includes about 1000 chemicals (covering several product names), possible to add new ones (since Feb 2009); physicochemical data compiled from several databases</td>
<td>Includes almost 600 chemicals (covering several product names and ‘vague’ contaminants, e.g. ‘coke oven emissions’), not possible to add new chemicals</td>
<td>Includes a database of 120 chemicals; a separate module, updating along with the new software version if needed; possibility to add chemicals; physicochemical data compiled in Dutch studies</td>
<td>Includes a database of 345 chemicals; a separate workbook, updating along with the new software version if needed; possibility to add chemicals</td>
<td>Includes 21 individual PAHs and PHCs + 11 PHC fractions (data in the workbook, updating along with a new software version if needed); not possible to add own chemicals (password protected)</td>
</tr>
</tbody>
</table>

<sup>a</sup>Version used in this study and also used to derive the most recent Finnish soil quality benchmarks (Ministry of the Environment, 2007); a new modified version is currently available.  
<sup>b</sup>SSL = soil screening level, the maximum concentration in soil that corresponds with the acceptable level of health risks;  
<sup>c</sup>Pathway-, exposure route-, and contaminant-specific limitations identified in the current research, see details in the text. The studied tools might have additional limitations when used to assess contaminants other than those covered in this research;  
<sup>d</sup>A new version (2.0) of this software was recently submitted, which includes updates to some parameters, e.g. in accordance to the studies by Rikken et al. (2001).
### 3B. Components included in risk calculations and documentation

<table>
<thead>
<tr>
<th>Feature</th>
<th>RAIS Chemical calculator</th>
<th>SSL calculator</th>
<th>Risc Human 3.1</th>
<th>CalTOX 4.0 (beta)</th>
<th>SOILRISK 1.0</th>
</tr>
</thead>
<tbody>
<tr>
<td>Exposure scenarios</td>
<td>6 (resident, inside/outside/worker, farmer, recreator); exposure routes within these specified, changes are not possible</td>
<td>any, defined by selecting suitable exposure routes, but not all routes are covered (see below)</td>
<td>any, defined by selecting suitable exposure routes</td>
<td>any, defined by building suitable scenarios from altogether 23 exposure routes</td>
<td>2 (residents, workers), exposure routes within these scenarios defined</td>
</tr>
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<td>Contaminant distribution</td>
<td>based on partition and diffusion coefficients, (volatilization)</td>
<td>based on partition and diffusion coefficients, and mass limit</td>
<td>based on the fugacity theory</td>
<td>based on the fugacity theory; possibility to study change in concentration over time (GW)</td>
<td>based on partition and diffusion coefficients</td>
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<td>+ (complete mixing)</td>
<td>+ (complete mixing)</td>
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<tr>
<td>- groundwater</td>
<td>-</td>
<td>+</td>
<td>-</td>
<td>+ (complete mixing)</td>
<td>+</td>
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<tr>
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<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
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<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
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<td>- surface water</td>
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<td>-</td>
<td>-</td>
<td>+</td>
<td>-</td>
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<tr>
<td>- plants</td>
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<td>+</td>
<td>+</td>
<td>+</td>
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<td>+</td>
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<td>- soil, dermal contact</td>
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<td>+</td>
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<td>- food</td>
<td>+ P, M, Mt, Fi</td>
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<td>+ S, W</td>
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<td>+</td>
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<td>- tap water, inhalation</td>
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<td>- other</td>
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<td>based on partition and diffusion coefficients</td>
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*Note:* + indicates presence, - indicates absence.
Table 3B continues

<table>
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<th>Feature</th>
<th>RAIS Chemical calculator</th>
<th>SSL calculator</th>
<th>Risk Human 3.1</th>
<th>CalTOX 4.0 (beta)</th>
<th>SOILRISK 1.0</th>
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<tr>
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<td>Both carcinogenic and non-carcinogenic response, separately</td>
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<td>Both carcinogenic and non-carcinogenic response, separately</td>
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<tr>
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<td>Exposure route-specific (RfD, RfC), quality standards for groundwater (transport), based on IRIS database (maintained by USEPA)</td>
<td>For total intake (TDI) and concentrations in air (TCA); references to data sources presented</td>
<td>Exposure route-specific (RfD, RfC) and for total intake (ADI, unit risk); sources described</td>
<td>Exposure route-specific (RfD) and unit risks for total exposure; data sources described</td>
</tr>
<tr>
<td>Outputs</td>
<td>1) CDIs, and corresponding hazard quotients and cancer risks for total exposure and for separate pathways (soil, SW, air, GW, food); children and adults not separated</td>
<td>1) Exposure route-specific SSLs 2) age-adjusted exposure 3) separate SSLs based on carcinogenicity and other health effects</td>
<td>1) Daily dose through different exposure routes and total risk as a hazard quotient (HQ) 2) Separate daily doses for adults, children, and a combination of both 3) concentration in different environmental and exposure media, fluxes (soil air)</td>
<td>1) Risk estimates (= excess cancer risk or hazard quotient) based on total exposure 2) Soil target conc. in different soil layers 3) Concentrations in different environmental media (different soil layers, GW, SW, air, sediment, plant leaves)</td>
<td>1) Soil target concentrations and 2) their exceedance (%)</td>
</tr>
</tbody>
</table>

2) Exceedance of the calculated site-specific target concentrations (%)

S = sediment, SW = surface water, W = water, Vo = volatiles, LV/RL = leafy/root vegetables, F = fruit, C = cereals, P = plants in general, M = milk, Mt = meat, Fi = fish, E = eggs, TDI = tolerable daily intake, TCA = tolerable concentration in air, CDI = chronic daily intake, GW = groundwater

*Transfer from soil and irrigation water, in the latter case concentration in water has to be given as the input; **Migration from irrigation water (surface or groundwater) to food crops; ***Including only migration from soil water to domestic water through pipe walls, if the groundwater at the site is used as a source of tap water, the concentration of contaminants in this water needs to be known. In the updated version (3.2.) consumption of groundwater used as drinking water has been added; *Exposure to groundwater which is not used as domestic water; *Only substances with assumed additive joint toxic action: PAH (10 compounds), some pesticides (carbofuran+carbaryl and drins = aldrin, dieldrin, endrin), some metals (Cd, Pb, Hg), some volatile chlorinated hydrocarbons (7 compounds), chlorobenzenes (6 compounds), chlrophenols (5 compounds), volatile aromatic hydrocarbons (9 compounds); *Option to conduct sensitivity analysis; **Compatible with Excel-based statistical software tools such as CrystalBall™, @Risk and ModelRisk; ***additivity assumed for 1) aliphatic PHCs, 2) aromatic PHCs, 3) BTEX, and 4) carcinogens
4.3.2 Determination of contaminant distribution and transport

In the studied multimedia HRA tools, determination of the distribution of contaminants between different environmental compartments is based on generally accepted approaches, i.e. the fugacity theory or empirical equilibrium partition coefficients. However, significant differences exist in the consideration of the soil structure and distribution of contaminants (whether heterogeneity in the soil structure and contamination are considered) and in transport algorithms (Table 3B). Some models also consider decay and transformation of contaminants. The physicochemical parameters and reference values vary in different tools, since they use different databases and data sources. SOILIRISK, being the only HRA tool developed in Finland, includes some default values corresponding to Finnish conditions. In the studied multimedia tools, the limitations of contaminant distribution and transport models are mainly associated with inbuilt QSAR models, which are very simple and fail to consider several contributing factors and the specific characteristics of some contaminants (see Section 4.2.2).

The organic carbon content (OC) of soil is a key parameter in multimedia models, since organic carbon can adsorb contaminants, thereby making them less mobile. It is therefore a key contributor to contaminant distribution in soil, and significantly affects the calculated contaminant concentration in soil water, and consequently, plant uptake (see the ‘Plants’ section below). Site-specific OC values should therefore always be preferred over generic values, particularly in the higher tiers of RA. In the studied HRA tools, the default values for OC vary from 0.6% (RAIS and SSL) to 5.8% (corresponding to 10% OM, Risc-Human). CalTOX, on the other hand, has an abundant choice of default values representing separate soil layers (upper soil, vadose zone, and aquifer) in 65 different regions (states) corresponding to different soil types. SOILIRISK uses the value of 1%, which was also adopted in developing the most recent Finnish soil quality benchmarks (Reinikainen, 2007). Looking at the geochemical data, till is the most common soil type in Finland, representing almost 70% of all soils, and sandy till is the dominant till type (75% of all till soils) (Koljonen, 1992). The subsurface soil (> 0.2 m) data on sandy till collected by the Finnish Environment Institute (altogether 1941 samples taken around the country) was compiled and fitted into a lognormal distribution curve using statistical SPSS software (Sorvari, 2000). This resulted in a mean value of 1.2% organic matter (equivalent to approximately 0.7% of OC), the maximum being 10.3% and the minimum, 0%. Using the mean OC value would result in more conservative risk estimates than using the default values included in the studied RA tools (excluding CalTOX), since the lower the OC, the higher the concentration of an organic contaminant in soil water (and in plants) and in air.

Air

All the studied HRA tools use diffusion coefficients to calculate contaminant distribution between soil and air or water and air, but differences exist in calculating contaminant mixing in air and distribution between indoor and outdoor air. Thus, the dispersion models, methods for calculating volatilization (infinite or finite source), and transport mechanisms involved (whether transport through cracks is considered or not, whether only diffusion or also advection is considered) vary (Rossi, 1999 & 2002; Chang et al., 2004; Poletti et al., 2004). Some vapor intrusion models (i.e. models used to determine contaminant transport to indoor air), such as the RBCA model, include highly simplified algorithms that consider only diffusion, while others include pressure driven flow induced by both wind and temperature difference (Ferguson et al., 1998). In practice, several factors contribute to contaminant transport from soil to air, including soil cover (e.g. clean soil, snow, asphalt, vegetation, and buildings), soil type, particle size, land forms, wind speed and direction, climatic conditions, season, and activities on site. Transport from soil to indoor air is also affected by pressure difference (outside vs. inside the building). In addition, volatilization is affected by
temperature, humidity, and air pressure. The default values of the key parameters, such as transfer factors (soil air-indoor air), intrusion factors, and meteorological data vary in different multimedia tools. According to Ferguson et al. (1998), many of the key parameters, such as soil permeability, air exchange rates in crawl spaces, and floor leakage rates, are hard to determine experimentally, making validation of soil vapor intrusion models difficult.

None of the studied multimedia tools consider all contributing factors or particle size, which is one of the key variables in the assessment of contaminant transport along with dust, since it determines transport distance. Risc-Human, on the other hand, includes a factor for considering any obstacles (land forms), but no variable that takes into account land cover (like RAIS), since the concentration of suspended particles in air is given as an input. The other studied multimedia HRA tools calculate dust concentration starting from concentrations in soil and using meteorological data.

Compared with other studied multimedia HRA tools, Risc-Human applies more variables related to the properties of the building in determining indoor air concentration and therefore, enables a more detailed assessment, if adequate data are available. It includes algorithms for two types of buildings: with a crawl space or with a basement, whereas neither the SSL nor the RAIS calculator separates indoor and outdoor air and therefore, no information about the building is needed. Due to this simplification, the uncertainty of the indoor air estimate of volatiles increases. SOILIRISK, on the other hand, enables considering the proportion of building (area) that is in contact or above contaminated soil and groundwater. The structure of the house, i.e. whether it has a crawl space, a basement or neither of these, is in fact an important factor in the selection of a suitable model for calculating exposure to indoor air, but usually unknown when dealing with future constructions. The structure of houses varies in Finland and ventilation practices (e.g. rate) can differ from those used in other countries. No universal transport algorithm therefore exists which would be suitable for assessing contamination of indoor air in all situations.

RAIS calculates volatilization only for chemicals exceeding specific limits for the factors governing volatilization (Henry’s law constant \( \geq 1 \times 10^{-5} \text{ atm-m}^3 \text{ mol}^{-1} \) and molecular weight \( \leq 200 \text{ g mol}^{-1} \)). Therefore, volatilization of DDT and lindane, which were identified as COPCs at the studied forest nurseries (Paper II), would be ignored in calculating human exposure using the RAIS tool. Calculations using Risc-Human proved, however, that inhalation of these COPCs had only a minor contribution to overall human exposure, and so their exclusion from RAIS is justified. Since there were no monitoring data available on the actual concentrations, it was not possible to verify the validity of the calculated COPC concentrations in air. Nevertheless, Poletti et al. (2004) and Walden (2005b) state that overall, the models inbuilt in multimedia HRA tools generally overestimate the actual contaminant levels in indoor air.

Additional studies, not included in this research, showed that Risc-Human (version 3.1) does not consider the saturation limit of contaminants when assessing their volatilization. As the example calculation in Table 4 shows, this can lead to significant overestimation of contaminant concentration in air, and consequently, the risks associated with exposure through inhalation. Other studied multimedia HRA tools consider saturation limits and show a warning in case of exceedance or indicate the limits along with the results.
Application of Risk Assessment and Multi-Criteria Analysis in Contaminated Land Management in Finland

Groundwater

The algorithms used to assess groundwater transport in multimedia tools differ in terms of the type of source they consider (finite source vs. infinite source), calculation of mixing zone depth and dispersion (dimensions), and the assumed distance to the receptor point (i.e. the point of compliance, POC). Some models also consider contaminant retardation caused by sorption processes, and decay.

Both SSL and Risc-Human failed to reliably assess the transport of contaminants from soil to groundwater. The SSL calculation tool uses simple QSAR models based on partition coefficients ($K_d$) and dilution factors, i.e. the same methods adopted in manual calculations (see section 4.2.2). The alternative algorithms included in the SSL calculator, on the other hand, turned out to be impractical since they ignore all contaminant properties, which can affect the sorption of contaminants onto soil particles. Risc-Human (version 3.1), in turn, does not include algorithms for assessing contaminant transport in groundwater. Risc-Human therefore proved to be unsuitable for assessing health risks related to contaminated groundwater used as drinking water when data on the contaminant concentrations in the former are missing (Paper II). In such a case, Risc-Human only assesses contaminant transport from soil to drinking water resulting from diffusion through water pipes, i.e. it assumes that local groundwater is not used for water supply. This can lead to significant underestimation of health risks, since private wells are rather commonly used for providing household water in Finland.

The RAIS Chemical calculator does not include algorithms for calculating contaminant transport from soil to groundwater. Of the surveyed software, only SOILIRISK and CalTOX considered that the source is finite, i.e. the concentration of a contaminant in soil declines due to groundwater transport and degradation. The SSL and RAIS calculators do not consider the distance to the POC, such as waterworks or the nearest well used to supply household water. Such simplifications are expected to lead to overestimation of risks, making these tools more conservative than the other studied HRA tools. All studied multimedia tools assumed complete mixing of contaminants in groundwater, although in CalTOX this only applies to the vertical direction (horizontal diffusion is considered). Such models are unsuitable for assessing non-aqueous phase liquids (NAPL), which form a separate phase in groundwater. Specific models should therefore be used for these contaminants. This is a significant shortcoming of most multimedia HRA tools, taking into account that more than 70% of contaminated sites in Finland involve petroleum hydrocarbons (Finnish Environment Institute, 2009a), which commonly involve NAPL.

<table>
<thead>
<tr>
<th>TPH fraction</th>
<th>$C_1$ mg/kg</th>
<th>$C_2$ mg/m$^3$</th>
<th>$L$ mg/m$^3$</th>
</tr>
</thead>
<tbody>
<tr>
<td>aliphatics &gt; C5 - C6</td>
<td>15</td>
<td>0.63</td>
<td>18.4</td>
</tr>
<tr>
<td>aliphatics &gt; C12-C16</td>
<td>$4.9 \times 10^3$</td>
<td>$1.9 \times 10^5$</td>
<td>1.0</td>
</tr>
<tr>
<td>aliphatics &gt; C16-C35</td>
<td>$4.9 \times 10^5$</td>
<td>$4.2 \times 10^8$</td>
<td>NA</td>
</tr>
<tr>
<td>aromatics &gt; C5-C7</td>
<td>0.53</td>
<td>0.16</td>
<td>0.4</td>
</tr>
<tr>
<td>aromatics &gt; C12-C16</td>
<td>$2.0 \times 10^3$</td>
<td>$7.8 \times 10^5$</td>
<td>0.2*</td>
</tr>
<tr>
<td>aromatics &gt; C21-C35</td>
<td>$1.6 \times 10^7$</td>
<td>$1.7 \times 10^{10}$</td>
<td>NA</td>
</tr>
</tbody>
</table>

*limit value for fractions > C8-C10, > C10-C12, and > C12-C16

Table 4. Example of the effect of contaminant saturation limit in the calculation of contaminant transport from soil to air: the concentration of different total petroleum hydrocarbon (TPH) fractions in respirable air and the corresponding limit value for protecting human health ($L$). $C_1 =$ concentration when the saturation limit is not considered, $C_2 =$ concentration when the saturation limit is considered. The original concentration in soil = 15002 mg/kg. (modified from Sorvari, 2006).
Plants

To derive plant concentrations from soil concentrations, multimedia HRA tools include algorithms based on either empirical studies (particularly models for metals) or QSAR models (organic chemicals) (see Section 4.2.2). Rosenbaum (2006) and Franco et al. (2007) identified significant disparities in these algorithms and default values used in determining contaminant transfer to plants in the multimedia tools they studied, e.g. in the approaches (whether transfer into stem uses steady-state assumption or equilibrium partitioning) and in volatilization rates. Both studies therefore suggested a careful review of the algorithm adopted in the modeling of transfer to plants.

Risc-Human separates root and stem vegetables in calculating contaminant uptake by plants (and consequent human exposure). The calculation of contaminant transport to leafy vegetables includes both root uptake and dry deposition (air particles), while only the former transport route is considered in the case of organic contaminants. Root uptake is based on soil-plant (metals) or soil water-plant (organics) bioconcentration factors (BCF). If no BCFs are available, the software uses QSAR models based on $K_d$ (inorganics) or $K_{ow}$ (organics) values to calculate the BCFs. In the case of organic contaminants, the calculation is very sensitive to the organic carbon content of soil, since the algorithm for calculating contaminant concentration in soil water uses soil OC as one variable (see Paper II). The concentration in plants therefore increases when OC decreases. In addition, the changes of $K_d$ values of metals along the variation in soil pH has not been taken into account in either Risc-Human or the other studied multimedia tools. Several models have been developed, however, which include soil properties such as pH and calcium content as variables affecting the plant uptake (e.g. Ryan et al., 1988; Bechtel Jacobs, 1998; Versluijs and Otte, 2001). Furthermore, more complicated models consider translocation of contaminants to other plant parts, and the effect of air-leaf exchange, metabolism and dilution by plant growth on contaminant concentration in plant (Collins et al., 2006).

The transport pathways for plant uptake included in the other studied multimedia HRA tools differ from those of Risc-Human. RAIS involves uptake from irrigation water in addition to root uptake, while CalTOX includes also rainsplash and uptake from soil and air and uses calculations based on the fugacity model. However, it is worth noting that in the case of metals, the BCFs used in Risc-Human implicitly also consider all uptake routes, since they are based on experimental studies. As shown in Table 3, the SSL tool and SOILIRISK do not include calculation of plant uptake of contaminants, owing to the fact that they do not consider food consumption.

According to the studies by Rikken et al. (2001) and Poletti et al. (2004), Risc-Human generates higher plant uptake estimates than other equivalent European software tools. In the studies of Finnish forest nurseries (Paper II), the effect of the plant uptake algorithm on the resulting human exposure to pesticides via edible plants was studied by entering manually calculated19 pesticide concentrations in plants to the Risc-Human program. In the case of two pesticides, Risc-Human generated exposure estimates which were 5- to 15-fold higher than estimates based on manually calculated plant concentrations. In the case of three pesticides, the results were reversed, i.e. Risc-Human produced lower estimates, the difference being 50-fold, at the maximum (Jaakkonen and Sorvari, 2006, Table 13). Differences in the $K_{ow}$ values did not explain these results.

In this research, some contaminant-specific limitations of plant uptake algorithms came up, namely the unsuitability of Risc-Human for determining plant uptake of very lipophilic contaminants, such as pesticides (Paper II). The problem of the poor predictability of QSAR models in the case of very lipophilic organic substances has in fact been identified in several previous studies (Suter et al., 2000; Rikken et al., 2001; Collins et al., 2006). Furthermore, Rosenbaum (2006) pointed out the risks of using a QSAR model based on $K_{ow}$ outside its valid range in assessing

19 Using QSAR models by Topp et al. (1986) for root uptake (tubular plants) and Travis and Arms (1988) for above-ground plant parts (leafy vegetables)
contaminant transfer to cattle by proving that failure to acknowledge this issue can result in unrealistic concentrations in beef and milk.

4.3.3 Calculation of human exposure

The exposure scenarios included in the studied multimedia tools to conduct HRA are summarized in Table 3B. The SSL calculation tool is the most incomplete, since it excludes several exposure routes, food consumption being perhaps the most important. Risc-Human and CalTOX differ from the other studied HRA tools in that they include a more diverse set of exposure times (e.g. separation of seasons, consideration of bathing and showering time).

Different multimedia HRA tools use different default exposure parameters, and the methods used to calculate human intake also vary. The former variation covers all exposure routes and occurs in fractions of contaminated media and averaging time, among others. Additional dissimilarities exist in the variables involved in specific exposure routes. In the study by Swartjes (2002) investigating seven European HRA software programs, disparity in calculation methods rather than variation in input parameters contributed to the high variability of dose estimates associated with crop consumption and inhalation of indoor air. The variation proved to be highly dependent on the contaminant.

The key elements, i.e. algorithms and default values, of the different exposure routes in the studied multimedia tools are briefly discussed below.

Soil ingestion and food consumption

Calculation of soil ingestion and food consumption is based on identical, universal equations (see Paper II, V and VI) in all the studied multimedia tools. Thus, differences only exist in the default values and whether or not the share of contaminated medium and contaminant absorption from it (bioavailability) are considered.

In the study by Swartjes (2002), no correlation was found between variation in exposure from vegetables and concentration in vegetable or soil water. Swartjes concluded that other variables, i.e. crop consumption and fraction of homegrown (contaminated) vegetables contribute to the variation. These parameters also varied in the multimedia tools examined in this research (Table 5). A majority of the studied tools take into account the contribution of home-grown or contaminated vegetables to the total consumption of vegetables, whereas none of them consider the effect of food preparation (washing, cooking) on removal of contaminants. Moreover, they ignore the fact that contaminant absorption from soil, food, or drinking water is usually less than 100%.

In the studied multimedia HRA tools, the default values of daily soil ingestion were of the same order of magnitude (Table 5). The estimates of soil ingestion values vary in different literature sources, mainly due to the use of different tracers, but also due to different study protocols. No data are available on appropriate Finnish values. A recent study involving both children and adults arrived at average intake estimates between 23 and 625 mg day\(^{-1}\) for adults and 37 and 207 mg day\(^{-1}\) for children, depending on the tracer\(^{20}\) (Davis and Mirick, 2006). It is worth noting that the children’s age in this study varied from 3 years to 8 years, whereas other studies and HRAs usually separate the age group of 1-6 years due to its common hand-to-mouth behavior, which leads to higher ingestion rates compared with adults. Nonetheless, it can be concluded that the default values included in the studied multimedia tools mainly represent rather conservative values.

Consumption of drinking water is the key parameter in assessing exposure to contaminants through household water. Based on Finnish statistics, among Finns the consumption varies from 0.33 to 4.0 l day\(^{-1}\), 1.5 l day\(^{-1}\) being the average value and covering both adults and children (Män-
nistö et al., 2003). Thus the value used in the calculations in this research (2 l day\(^{-1}\)) corresponds to a rather realistic estimate, which also equals the figure commonly adopted in the multimedia HRA tools.

**Dermal intake**

Calculation of dermal intake in Risc-Human differs from that of the other studied multimedia tools. While RAIS, CalTOX, and SOILIRISK all use absorption factors to determine dermal exposure to contaminants in soil, Risc-Human uses permeability coefficients and assumes that dermal intake of metals and other inorganic chemicals is insignificant. The SSL calculator totally ignores dermal exposure. The limitations of the method based on permeability coefficients were already discussed in Section 4.2.3. Some HRA tools not examined in this research apply mass balance calculations in assessing dermal exposure (Poletti et al., 2004). Such models are based on the assumption of conservation of mass (input = output), and they determine the contribution of dermal intake on the basis of mass flows.

Both the permeability coefficient and the absorption factor method require data on the amount of soil adhered to the skin. Again, the default values in different multimedia HRA tools vary. Permeability coefficients as well as absorption factors are chemical-specific parameters, but still, some tools use a constant across all chemicals for the latter variable (see Table 5). The risk assessments in this research only showed a minor contribution of dermal intake to overall human exposure at the study sites (Paper II). Alternative methods for assessing dermal absorption were therefore not examined in more detail.

**Inhalation**

Excluding the SSL and RAIS calculators, the studied multimedia HRA tools separate indoor air and outdoor air in assessing exposure through inhalation. All of them make a difference between volatiles and particles. The exposure algorithms are basically similar, differences occurring mainly in variables, such as inhalation rates and lung retention factors. The results from the study by Swartjes (2002) showed no distinct correlation between variation in concentration in soil air and variation in calculated human exposure to indoor air. These results suggest dominance of the variability of algorithms applied in calculating contaminant mixing in air and distribution between indoor and outdoor air, as well as the number and type of variables describing the properties of the building.

The fraction of respirable dust in air is relevant from the viewpoint of human exposure, since it consists of particles that can penetrate deep into the lungs (size < 10 µm). As mentioned in section 4.3.2, particle size of soil is not explicitly considered in the studied multimedia HRA tools, although Risc-Human includes variables describing absorbed fraction and fraction retained in lungs, both of these depending on particle size. Both of these variables thus contribute directly to the final risk estimate associated with exposure through inhalation. For example, changing the value of absorbed fraction changes the inhalation dose estimate by the same factor. However, since no adequate data on these variables were available, the default values included in Risc-Human were adopted.

Inhalation (of indoor air) was a significant exposure route only in the case study dealing with forest nurseries (Paper II). It turned out that Risc-Human and the SSL calculator can produce controversial results. For a couple of pesticides, the latter generated SSL values for inhalation route that were lower than the SSLs associated with soil ingestion, thereby showing higher risks associated with the former exposure route. Whereas the results from HRA using Risc-Human showed that soil ingestion and in most cases even dermal intake exceeded exposure via inhalation. These results suggest that the SSL tool is more conservative in calculations associated with exposure through inhalation.
Summary of the key parameters involved

Table 5 summarizes the default values of the key exposure parameters in the studied multimedia tools for HRA, from the viewpoint of the case studies included in this research, and the corresponding values used in the site studies. The latter figures vary slightly due to the different level of conservatism of the HRAs. CalTOX uses different methods and partly different parameters, which complicates its comparison with other tools. The differences in the values of ingestion of vegetables seem to be the highest. Here, the low value of the SSL calculator is compensated by the higher value it uses for the variable ‘Fraction of contaminated vegetables’. The food consumption rates used in this research were compiled from Finnish food statistics. These statistics did not include an explicit distinction between children and adults, which creates some uncertainty in the adult/child figures.

Table 5. Default values for some of the key exposure parameters in the studied multimedia tools and the corresponding values used in this research. SSLc = SSL calculation tool, R-H = Risk-Human, GW = groundwater

<table>
<thead>
<tr>
<th>Exposure parameter</th>
<th>RAIS calculator</th>
<th>SSLc</th>
<th>Risc-Human</th>
<th>CalTOX</th>
<th>SOILI-RISK</th>
<th>This research</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil ingestion mg d⁻¹</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>default SSLc &amp; R-H</td>
</tr>
<tr>
<td>child</td>
<td>200</td>
<td>200</td>
<td>150</td>
<td>60</td>
<td>-</td>
<td>25</td>
</tr>
<tr>
<td>adult</td>
<td>100</td>
<td>50</td>
<td></td>
<td>10</td>
<td>10</td>
<td>100</td>
</tr>
<tr>
<td>Intake, drinking water l d⁻¹</td>
<td>1.0</td>
<td>1.0</td>
<td>calculated as a fraction (0.8 for GW) from water use</td>
<td>1.0</td>
<td>2.0</td>
<td>default R-H</td>
</tr>
<tr>
<td>child</td>
<td>2.0</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2.0</td>
</tr>
<tr>
<td>adult</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ingestion, vegetables g-fw d⁻¹</td>
<td>10.4</td>
<td>76.1</td>
<td>varies by age</td>
<td>4.16</td>
<td>-</td>
<td>180</td>
</tr>
<tr>
<td>child, leafy/tuberous</td>
<td>28.5</td>
<td>74.8</td>
<td></td>
<td></td>
<td></td>
<td>110</td>
</tr>
<tr>
<td>adult, leafy/tuberous</td>
<td>1</td>
<td>0.1</td>
<td></td>
<td>0.47</td>
<td>-</td>
<td>0.1</td>
</tr>
<tr>
<td>fraction of contaminated vegetables</td>
<td>0.1</td>
<td>0.51</td>
<td></td>
<td>0.52</td>
<td>default R-H</td>
<td></td>
</tr>
<tr>
<td>Dermal intake, soil adherence mg cm⁻²</td>
<td>0.2</td>
<td>0.52</td>
<td></td>
<td>0.5</td>
<td>0.52</td>
<td></td>
</tr>
<tr>
<td>child</td>
<td>0.07 t</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>adult</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Absorbed fraction</td>
<td></td>
<td>chemical-specific</td>
<td>0.15</td>
<td>0.2</td>
<td>0.06; default R-H</td>
<td></td>
</tr>
<tr>
<td>Inhalation breathing volume m³ d⁻¹</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>child</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>adult</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>worker</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fraction retained in lungs</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: *age-adjusted value = 114 (childhood 6 years, adulthood 64 years); the multimedia tool does not use this variable in calculating risks for this exposure route; for example the SSL calculator only compares calculated groundwater concentrations against drinking water standards, and therefore, it does not use data on ingestion; ' outdoors; 'indoors; ‘calculated from the value 2.2 x 10⁻⁴ kg kg⁻¹ d⁻¹ (Rikken et al., 2001); 'calculated from the value 1.4 x 10⁻⁷ kg kg⁻¹ d⁻¹ (Rikken et al., 2001); 'age > 19 years, includes fruits and vegetables; 'Paper II; 'Paper V; 'Paper VI; 'Paper V, calculation using a biokinetic model
4.3.4 Other transport pathways and exposure routes

Potential surface water contamination and the consequent risks to aquatic ecosystems and human health was an issue in two case studies included in this research (Paper II and V). However, owing to the scope of these studies (land contamination), only tier 0 level ERA, i.e. comparison of contaminant concentrations in surface water with ecological benchmarks of some aquatic key organisms, was conducted. The lack of site data (Paper II) and the limited size of the water body (Paper II and V) also justified the exclusion of a higher tier ERA. The water bodies were actually small ponds which were therefore not expected to support receptors of higher trophic levels, and the contamination of which would – due to its small size – not pose ecological risks that would be significant in spatial scale.

Risc-Human, RAIS and CalTOX include a component for assessing human exposure through fish consumption. Only CalTOX involves algorithms for calculating contaminant concentration in surface water (via runoff, i.e. only soil erosion considered), whereas the concentration in water phase needs to be entered as input for the other studied multimedia tools. Risc-Human can alternatively use the measured concentration in sediment and sediment properties for calculating the concentration in water, using partition coefficients. Calculation of contaminant concentration in fish is based on using water-fish bioconcentration factors in all the studied multimedia tools. Since bioconcentration in fish is determined on the basis of the soluble contaminant fraction, the method is poorly applicable to the assessment of risks related to fish consumption caused by COPCs with low solubility and high affinity to the sediment, such as dioxins, particularly in a case where fish consumption is focused on bottom feeders (such as flounder). In the case of such contaminants, an alternative method should be used or preferably, bioconcentration in fish should be determined by other means (e.g. bioassays).

Risc-Human, RAIS, and CalTOX also consider additional food items, such as meat and eggs (see Table 3B). The algorithms involved are based on biotransfer factors and determining the exposure of livestock and poultry (CalTOX). Exposure through other food items, such as fish, beef, milk, and eggs, was not relevant at any of the study sites included in this research.

4.3.5 Dose-response assessment

All the studied multimedia HRA tools calculate risk estimates using the dose versus safe dose or dose cancer slope factor equations. Unlike the other studied tools, Risc-Human does not make any difference between carcinogens and non-carcinogens, meaning that the safe doses included in it consider carcinogenicity. The averaging time in the calculation of cancer risk estimates is consequently different in Risc-Human, i.e. instead of the default lifetime applied for carcinogens it uses the same exposure time for both carcinogens and non-carcinogens. However, it is often useful to separate the different endpoints, i.e. cancer and non-cancer health effects. Since development of cancer requires a long exposure time, typically decades, reference values based on this endpoint are not suitable for use in cases with shorter exposure times.

The acceptable cancer risk levels vary in the different multimedia HRA tools: the SSL calculator, RAIS calculator, and CalTOX apply the value of $10^{-6}$, SOILIRISK the value of $10^{-5}$, and Risc-Human the value of $10^{-4}$. Selection of an acceptable risk level is a policy decision. Differences also exist in the acceptable daily intake values, i.e. ADIs/TDIs/RfDs. Similarly to soil benchmarks (Section 4.2.1), these differences can arise from various factors, and sometimes no consensus exists internationally even on toxicity mechanisms of particular chemicals. It turned out that in many cases it is difficult to track the bases or even the origin of the default reference values. At the same time, the variation in reference values can lead to considerable disparities in the final risk estimates (Paper II and V). This problem was also identified in the HRA software comparison.
studies by Swartjes (2002) and Poletti et al. (2004). It is worth noting that it is not possible to modify the reference values in many HRA tools (e.g. SSL, RAIS, and SOILIRISK). Therefore, in characterizing the risks it would be important to also study separately the dose estimates and alternative reference values, and the effect of the latter on the risk estimates. Ideally, the basis of the reference values should be studied in order to exclude clearly unsuitable ones, e.g. on the basis of exposure duration.

The multimedia HRA tools involved in this study do not consider background exposure from outside the contaminated site in the calculation of risk estimates. The reference values do, however, refer to total exposure where the background is included. Background can significantly contribute to total exposure (Paper V and VI), particularly in the case of contaminants that tend to bioconcentrate in food items, such as dioxins and some persistent pesticides. Smoking and occupational exposure can also increase substantially the overall contaminant intake. Surveys of previous Finnish risk assessments showed that background exposure has very rarely been considered even qualitatively in the characterization of risks (Sorvari and Assmuth, 2000; Sorvari, 2004c).

4.4 Specific issues associated with risk assessment

4.4.1 Climatic and regional factors

As presented in the preceding sections, algorithms used in determining human exposure include various parameters; some of these are broadly applicable in different parts of the world (e.g. human body weight), whereas others should be adjusted to correspond to the actual conditions at the study site. From the viewpoint of RA in Finnish conditions, the effect of climatic differences is worth noting since the RA tools were generally developed in countries with a temperate climate. From this it follows that some default values in multimedia HRA tools are inappropriate and need to be changed.

Firstly, about half of Finland experiences an annual snow cover period longer than 100 days (Finnish Meteorological Institute, 2006), and it is evident that during this time all exposure routes related to direct human contact or contact of biota with soil are non-existent. The exposure times associated with outdoor activities can also be shorter due to climatic reasons. Besides, the shorter growing period compared with more temperate climates affects the accumulation of contaminants in plants and the following secondary exposure of biota. It is also evident that species (plants, animals) can differ, which reduces the applicability of any uptake and bioconcentration models (plants, animals) developed abroad (see Paper V).

Volatilization and chemical reactions in general are slow at low temperatures, and therefore, the transport rates of contaminants from soil to other environmental compartments in cold climates are usually lower than in temperate climates. Moreover, while frost efficiently hinders any contaminant transport to groundwater in wintertime, on the other hand the freeze-thaw processes can change the soil structure by increasing void space and creating cracks which change the hydraulic conductivity of soil, and consequently enhance leaching of contaminants (e.g. Othman et al., 1994). Freeze-thaw cycles can also have other implications in contaminant mobility, which should be considered in risk management. For example, some studies suggest that they cause substantial remobilization and distribution of LNAPL plume (Niven and Singh, 2008). In spring, the rising groundwater table reaching the contaminated soil layer can further promote contaminant transport to groundwater. Changes in hydraulic conductivity can be taken into account by selecting an appropriate input value for this variable, while the effect of temperature differences can be considered by adjusting the relevant physicochemical parameter values (e.g. reaction rates). It is however, evident that simple multimedia models fail to consider exceptional conditions such as a rising groundwater table and its effect on migration of contaminants. The heterogeneity of Finnish soil further complicates the determination of contaminant transport in soil (Kuusela-Lahtinen and Vahanne, 2005).
Climatic factors also affect degradation of chemicals in the environment. Several studies have indicated that decay rates of organic chemicals in Finnish soil and groundwater are generally much lower than reported in the international literature (Paper II). This needs to be considered when assessing the time span of risks.

Finally, it is clear that there are also additional regional and demographic differences that should be considered when adopting models developed abroad. Therefore, data representative of Finnish conditions available in different statistics should be systematically compiled for use in future RAs.

4.4.2 Mixture effects

The surveys of previous Finnish risk assessments (Sorvari and Assmuth, 2000; Sorvari, 2004c) showed that mixture effects, other than those following concentration/dose additivity (e.g. dioxins), have generally been ignored and not even discussed as a potential issue. RA based on the use of multimedia models, as well as RA of chemicals in general, mainly focuses on individual substances. Some HRA tools, e.g. Risc-Human, consider combined toxicity, assuming concentration/dose additivity. In Risc-Human the following groups of chemicals are assumed to comply with the additivity theory: chlorobenzenes, chlorophenols, some pesticides and metals, PCBs, and PAHs. In this research, possible mixture effects, i.e. joint toxic actions, were an issue at all the study sites (Paper I, II, III, V, VI). The problem was the lack of data on such effects in the case of chemical mixtures involved at a particular site and the fact that site investigations were focused on chemical studies (i.e. no toxicity tests were run).

The results from the ecotoxicity tests described in Paper I might have indicated some joint toxic actions deviating from additivity or alternatively, the presence of additional contaminants that were not analyzed or other conditions unfavorable to the test organism, since the magnitude of toxic response did not correlate with the concentrations of COPCs in all the ecotoxicity tests. This proved the well known inadequacy of chemical methods in assessing ecological risks, particularly when multiple chemicals are involved.

4.4.3 Spatial and temporal scale of risks

Time span, i.e. the period over which risks are determined, is a crucial factor in risk assessment. In the case of human exposure, the multimedia HRA tools consider time span through the parameters “averaging time” and “exposure time”. There seems to be general consensus on the value of the former parameter and hence, a default value of 70 years in the case of lifetime cancer risks and 30 years in the case of other risks is generally applied. In addition to the averaging and exposure times, particularly the time span of contaminant transport is also a relevant factor, since it affects the concentration of a particular contaminant in the exposure/contact medium at the time of exposure. Of the studied calculation tools and software programs, only CalTOX and SOILIRisk include components for considering the time perspective of contaminant transport. Still, the question remains, what is a suitable time span to consider, particularly in the case of slow transport or bioconcentration, since no guidelines have been issued in this context. This also applies to ecological risk assessment. In the case study concerning shooting ranges (Paper V), the time span of the corrosion of ammunition, which can increase the amount of available contaminants in different environmental compartments (e.g. soil, groundwater, surface water, plants), is a specific issue that is not considered in any of the multimedia HRA tools studied. Degradation, in turn, decreases the original concentration of contaminants in the environment. Ignoring degradation can therefore result in overly conservative risk estimates.

Spatial scale is another factor that should be considered in RAs, but it is seldom explicitly considered in the simplest multimedia HRA tools. The dimensions of contamination should be
defined on the basis of site studies (monitoring of contaminant distribution) and/or modeling. These studies should also consider the time span of contamination, i.e. any future changes in contaminant distribution. From the viewpoint of risk management, the choice of a suitable point of compliance is thus important. What is to be protected affects the selection of a POC, for example whether the contaminant concentration in groundwater has to be below the quality standard right beneath the contaminated spot, at the waterworks, or in a neighboring site. Here again, the time scale should be decided. In Finland the discussion about the temporal and spatial scale of risks is still ongoing, and therefore no clear guidelines for setting these risk factors exist so far.

4.4.4 Practical aspects associated with certain contaminants

Adequate and suitable site studies that consider the needs of site-specific RA are the prerequisite for generating reliable risk estimates. In the studies involved in this research, some issues associated with chemical studies of specific COPCs came up. The identified problems reduce the accuracy of risk estimates. Efforts should therefore be made to minimize their effect in future investigations at contaminated sites.

In the case of petroleum hydrocarbons (PHC), a lack of analysis data that reflect risks to human health or biota was a key issue (Paper I and III). Chemical studies of sites contaminated by PHCs are typically focused on total petroleum hydrocarbon (TPH) and some ‘indicator’ compounds, such as BTEX and PAHs. It has been common practice in Finland to analyze only TPH and these chemicals. At the same time, neither benchmarks nor reference values are available for such a complex mixture as TPH, and consequently, it is not possible to derive risk estimates based on TPH concentration data. Combining TPH analysis with determination of indicator compounds still leaves most of the TPH unknown. This can lead to serious underestimation of risks. However, reference values and benchmarks exist for separate TPH fractions (see Paper I), and they should be analyzed in order to be utilized in HRAs.

Some problems related to chemical analysis was identified in the case of PCB compounds. Although the analytical methods for measuring PCBs are consistent, the practices used to interpret the results, i.e. to calculate the total PCB concentrations, vary. It was therefore pointed out that the method of deriving PCB concentrations needs to be documented in order to ensure consistency with the benchmarks and reference values used in determining risks (Hellman et al., 2003).

At shooting ranges, contamination of soil samples by small lead shot particles turned out to be a problem (so-called ‘nugget effect’) (Naumanen et al., 2002; Ref. in Paper V). The nugget effect leads to overestimation of COPC concentrations, which in turn can lead to unrealistic risk estimates. These problems could be minimized by increasing the sample size.

Surveys of previous site-specific RAs of Finnish sawmill areas contaminated by polychlorinated dioxins and furans (PCDDs and PCDFs) showed several shortcomings in the assessments (Assmuth and Sorvari, 1998 & 2003). PCDDs and PCDFs have often been treated only as toxic equivalent concentrations (TEQ) without adequately addressing the actual PCDD/PCDF composition. However, different congeners differ considerably from each other in terms of environmental fate, bioavailability, and toxicity. Using only TEQ concentrations can therefore lead to unrealistic or erroneous risk estimates. Since highly chlorinated congeners (≥ 5 chlorine atoms) and furans are the dominant chemicals in Finnish sawmill areas (e.g. Sorvari, 2001b), it is important to consider these congeners separately in RAs.

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21 A complex mixture refers to a mixture that is comprised of tens, hundreds, or thousands of chemicals, the composition of which is qualitatively and quantitatively not fully known, whereas a simple mixture consists of a relatively small number of chemicals (e.g. ten or less), the composition of which is qualitatively and quantitatively known. (Feron et al., 1998).
The chemical state and speciation of a contaminant significantly affects its environmental fate and toxicity; the difference in toxicity of different species of a single chemical can be several orders of magnitude (e.g. in the case of chromium). While multimedia models generally ignore chemical speciation as belonging to higher tier RA, some contaminants having different chemical forms or valences with significant differences in toxicity are, however, generally separated. Such chemicals at least include the different valences of arsenic (As$^{+3}$ and As$^{+5}$) and chromium (Cr$^{+3}$ and Cr$^{+6}$), and the organic and inorganic forms of mercury. The need to separate the different species in chemical analyses is seldom considered, though, often due to the analytical difficulties and high costs of such investigations. In practice, different species often occur simultaneously in the environment, and their occurrence and dominance also varies along with changing conditions. In such cases a conservative approach would call for using the most toxic form as a basis in RA.

4.5 Evaluation of the methods and tools for uncertainty analysis

The qualitative rating used in the uncertainty analysis as part of tier 1 HRA (Paper V) proved to be a feasible way to systematically study the reliability and conservatism of the final risk estimates. However, the difficulties of such rating involve combining all identified uncertainties to derive an overall uncertainty estimate. Furthermore, the problems related to qualitative rating systems in general (Section 4.2.1) also apply to rating of uncertainties. Thus, this method is mainly useful to identify the main uncertainty factors and the need for a more detailed RA, e.g. to show whether the calculated risk estimates are conservative enough to reach an adequate protection level in case they imply that no RM actions are needed; or alternatively, whether they exhibit a need for RM actions but seem to be overly conservative.

Monte Carlo simulation provided a means to quantitatively study the uncertainties of the HRA of PCB contaminated sites (Paper VI) and ERA of an abandoned landfill (Paper I). In the former study, incorporating statistical analysis into manual calculations using generic exposure models proved to provide useful information on the probability of the deterministic risk estimates and verified that a more detailed assessment was not warranted. The need for quantitative statistical tools is emphasized particularly in the case of more detailed models and calculations including a high number of variables, since it is difficult to aggregate the variability of all parameters using simpler methods, such as varying one-by-one the values of all variables involved. It needs to be noted, however, that the results of an uncertainty analysis cannot be accurate if the input data are flawed. Unfortunately, a lack of accurate statistical data for each key parameter is often a problem in statistical analyses. It is common to make assumptions regarding the form of distribution functions since it is not possible – or even sensible – to determine all the statistical numbers and distributions of all parameters involved within a site-specific RA. Literature was therefore used as the main source of statistical data in the study described in Paper VI. Assumptions were made regarding statistical distributions also in the uncertainty analysis related to the ERA described in Paper I. An additional data gap associated with statistical uncertainty analysis is the lack of data on correlations between parameters. It is therefore a common practice to ignore any correlations; the same approach was adopted in the case studies included in this research.

4.6 Usability of solubility tests, biomonitoring, and bioassays in RA

Laboratory-scale solubility and leaching tests, such as the column test (Paper V), provide a means to estimate leaching of contaminants in short term and/or long term (time span depends on the test). However, due to the heterogeneity of soil (varying physiochemical properties) and uneven distribution of contaminants in soil, several samples are needed if a realistic estimate of contaminant transport is required. In the study described in Paper V, two samples representing different soil
types were selected for the column leaching test in order to have an estimate of extreme values of leaching rates. This methodology proved to provide a wide range of leaching rate estimates, which were further used to define the time span of contaminant transport to groundwater in different soils.

According to the literature, the column test generally overestimates actual contaminant transport (Aalbers et al., 1996), probably because it fails to consider resorption of contaminants in deeper soil layers and because the liquid-solid (L/S) ratio used in the test is generally higher than in actual field conditions. The L/S ratio also varies in different seasons and weather conditions. It is also possible that the liquid is not evenly distributed in soil matrix in actual site conditions and therefore not all contaminant molecules would be in contact with it and inclined to dissolution. Disturbance of the matrix during sampling can also enhance the solubility of contaminants. The fact that the concentration of lead (7 µg l\textsuperscript{-1} at the maximum) was below the drinking water standard in the groundwater samples taken at the study site (Paper V), despite significant leaching shown by the laboratory tests (180-8400 µg l\textsuperscript{-1} depending on the soil type) (Naumanen et al., 2002), supports the statement of overestimation.

The column leaching test as well as simple agitation tests simulate a stable condition and therefore fail to consider any disturbances or changes in the soil structure (see Section 4.4.1) or exceptional environmental conditions, e.g. change in pH. Availability tests conducted in varying pH conditions can provide data on the effect of changing pH and the availability of contaminants in the worst case situation if the acidity of the soil changes. Such tests were therefore used to study the solubility of lead from the shooting range soil (Paper V). The results offer information on the maximum proportion of contaminants available, for example to plants by root uptake, but not to other organisms, however, since they can be exposed via routes other than soil water.

Solubility tests with mild solvents were used for determining the availability of metals in present site conditions. Comparing the result from such tests to the data on total concentrations also enables evaluation of the conservatism of the exposure calculations based on total concentrations (Paper I and V). In the study described in Paper I, around 65% of the metals, at the maximum, were dissolved when using ammonium acetate as a solvent, meaning that the metals were in fact only partially available.

Biomonitoring serves as a method to verify exposure and evaluate the conservatism of risk estimates determined using exposure models (Paper VI). Human biomonitoring has very seldom been adopted in studies related to soil contamination in Finland, probably due to the practical difficulties involved in such studies, such as finding volunteers and a reference group and the extensive need for resources and specific expertise. Most contaminated sites in Finland are also spatially limited, which often makes human biomonitoring studies unnecessary. Several confounding factors can bias the results and therefore, depending on the contaminant to be monitored, thorough background data need to be compiled concerning characteristics, lifestyle, and eating habits of the study population (e.g. Sorvari et al., 2007). In the study conducted as part of this research, biomonitoring verified that the calculated human intake estimates overestimated actual exposure (Paper VI). Finnish human biomonitoring studies in a former smelter area (Vantaan kaupunki, 2002) contaminated by lead and a sawmill site contaminated by dioxin (Kaakkois-Suomen ympäristökeskus, 2003) produced similar results. In the case of ERA the usefulness of biomonitoring is evident since it can produce more realistic risk estimates (Paper I).

4.7 Barriers to eco-efficient CLM and means to promote it

The study on present practices and possible barriers to eco-efficiency of CLM in Finland showed several shortcomings and development needs (Paper IV). Since efficient recycling of excavated soil was considered one of the key factors of eco-efficiency, a lack of adequate policy instruments, e.g. guidelines and quality standards for excavated soil, is a clear barrier to its realization. The lack of
guidelines also pertains to realization of the Best Available Technology (BAT) principle, which is one of the basic components constituting eco-efficiency or sustainability. Since costs play a crucial role in determining concrete RM measures and their dimensions, a lack of efficient economic instruments, too, hinders the realization of eco-efficiency. The exemption of soils from waste tax when delivered to landfills, in particular, has been criticized by many. In addition, several experts who contributed to the study mentioned the insufficiency of public funding and funding systems as a problem. From the viewpoint of risks, various problems with the application of RA methods and inadequate adoption of the results from RA in decision-making are the main contributors to the lack of truly risk-based land management in most cases. The survey of previous RA cases revealed that the problems in them extended to all RA stages, i.e. from selection of methods to risk characterization. Moreover, it is not only the technical issues associated with RA that should be considered when aiming for eco-efficient CLM, but also the risk communication aspects. Sufficient and timely risk communication and stakeholder involvement were recognized as a crucial element in the attainment of eco-efficiency.

The results of the study were utilized to focus the efforts of further studies within the PIRRE project (see Paper IV). It was concluded that a DSS that comprises all elements relevant from the viewpoint of CLM and its eco-efficiency should be developed. While such systems were already available in some countries (Table 1 in Paper IV), it was considered important to establish a system that is clearly designed for Finnish purposes and takes into account the specific needs of guidance, the practical decision framework, the types of contaminated sites, and administrative practices in Finland. An Internet-based DSS was therefore developed, which includes guidelines and data on risk assessment, cost estimation, assessment of environmental and social effects, risk communication, and public participation. The tool, known as PIRTU (see Section 3.1.2., 4.8.3. and 4.8.4.), is an essential element of this DSS.

4.8 Decision-making using MCA

4.8.1 Evaluation of the TRIAD approach

The TRIAD procedure applied in the ERA of a former landfill site (Paper I) proved to be useful, but also to include some pitfalls, which should be acknowledged when interpreting the results. There is a risk of relying on the results of mechanical calculation and ignoring their thorough analysis, similarly to many previous site-specific HRAs that applied multimedia software tools in Finland (Sorvari and Assmuth, 1999 & 2000; Assmuth and Sorvari, 2003). It is therefore important to critically review the results and consider the reliability of the results from different types of studies (Lines of Evidence, LoEs), and individual tests and studies involved. It also turned out that the problem of finding a reference site and sample can be a real issue if TRIAD is used to conduct ERA. In addition, hormesis can complicate the processing of the results from toxicity tests.

The previous case studies conducted abroad (see Paper I) using TRIAD in the context of land contamination have considered the results from separate LoEs equally important. However, since the reliability of the results from different methods of ERA can vary considerably due to, for example, differences in method-specific procedures, such as modifications of the soil matrix, it was found useful to weight the methods and aggregate the results using an MCA approach (Paper I). The benefit of such MCA is that many uncertainties that are not explicitly included in the uncertainty analysis can be quantitatively covered.

According to Swartjes et al. (2008), the poor implementation of TRIAD and other equivalent approaches in practical ERA in Europe is mainly due to the lack of a clear regulatory framework. In Finland, however, the limited number of megasites, i.e. large contaminated areas where ecological risks could be a real issue, may also hinder the use of this methodology in the future. In the
small sites prevailing in Finland, ecological risks are expected to be spatially quite limited and therefore more manageable than in the case of large sites. In such a case, risk managers can be reluctant to invest in ecological studies involved in the TRIAD approach, particularly since it can produce controversial results and results that are difficult to interpret, which complicates their use in practice (see Paper I). In addition, the number of experts available for conducting the assessment and interpreting the results is very limited in Finland. These factors result in the conclusion that the TRIAD approach will probably be applied in only a few cases.

4.8.2 Feasibility and usefulness of MCA in the context of TRIAD

In the study described in Paper I, the multi-criteria analysis technique was incorporated into the TRIAD procedure. The methods were rated using expert judgements and further ranked on the basis of specific criteria. The researchers responsible for planning and implementing the separate studies in each LoE, i.e. data experts (as detailed by Critto et al., 2008), executed the rating of the separate ERA methods, whereas the risk assessor acted as the system expert and conducted the weighting of the assessment criteria used in evaluating the methods. Exposure time was considered the most important assessment criterion, since the measurement endpoints should be able to depict the long-term effects of contamination of biota on site. Of the toxicity tests, both invertebrate tests (enchytraeids and earthworms) were ranked a bit higher than the plant tests. According to several studies, the weighting method can significantly affect the results (e.g. Pöyhönen and Hämäläinen, 2001). In this study, however, direct weighting and pair-wise weighting produced almost equal results, probably owing to the simplicity of the decision problem, i.e. the low number of criteria to be weighted and their comparability. In addition, all the factors involved were at the same level of hierarchy (no sub-criteria, i.e. attributes, were involved).

MCA proved to provide a feasible means to consider the performance of the different methods in predicting risks at the study site. To avoid any bias, it is important that the rating of methods and weighting of the decision criteria are conducted by appropriate experts in that particular field. The effect of the scales used in the rating and weighting was not examined in this study. In practice, different scales should not change the results of rating in this case, since the rates were normalized between the values 0 and 1 and therefore, only relative performance was considered. Using the two different weightings ensured that the final weights were in accordance with the system expert’s views.

4.8.3 Preferred risk management approaches identified by the PIRTU tool

The preferred risk management options for the two model sites (gasoline station and shooting range) were defined by first determining values for the decision criteria, attributes, and sub-attributes involved, and then calculating preference scores with the PIRTU tool on the basis of the values of each criterion, attribute, and sub-attribute and the weights assigned to them by several experts representing different stakeholder groups (Paper III). Costs, risk reduction, and environmental effects turned out to be important decision criteria in both study sites, whereas the contribution of the ‘Other factors’ criterion (involving public image aspects and adverse ecological impacts, i.e. loss of habitat and biota due to remediation) to the total preference score was only minor (Fig. 5).

Figure 5 shows that in the case of the gasoline station, the RM alternatives based on MNA were the most preferred. In practice, however, the uncertainty of risk reduction and the long time span of remediation can diminish the eco-efficiency of MNA. Calculations using Risc-Human resulted in low risk estimates associated with soil contamination already prior to any remedial actions. The reason for this was that, due to a lack of data, only the TPH concentration and the concentrations of some indicator contaminants were used in determining the risk estimates, i.e. the hazard
quotients (see also Section 4.4.4.). The resulting hazard quotients therefore did not reflect the overall risks at the study site.

A. Gasoline station. Alt 0 = no remediation, Alt Ia&IIa = composting and reuse, Alt Ib&IIb = landfill treatment, Alt Ic&IIc = combustion, Alt III = MNA, Alt IV = SVE+MNA; in Alt Ia,b,c remediation target = old soil limit values (= SAMASE values); in Alt IIa,b,c remediation target = old soil guideline values (= SAMASE values); in Alt Ia,b,c & IIa,b,c groundwater remediation using pump-and-treat (absorption into activated carbon).

B. Shooting range. Alt 0,V&VI = no soil remediation, Alt I&II = landfill treatment, Alt III = soil washing, Alt IV = topsoil removal & recycling of ammunition; in Alt I remediation target = old soil guideline value (= SAMASE value); in Alt II remediation target = new soil guideline value; groundwater management: Alt 0-III = building of a new waterworks, Alt IV = reactive barrier, Alt V = in waterworks using Metclean, Alt VI = treatment in waterworks using membrane filtration.

Figure 5. Comparison of the preference of soil (and groundwater) RM options for the gasoline station (A) and shooting range (B) on the basis of preference scores and the contribution of separate decision criteria to them. The highest score indicates the most preferred alternative (Paper III, © Elsevier Ltd, 2009). MNA = Monitored Natural Attenuation, SVE = soil vapor extraction.
In the case of the shooting range, the alternatives where groundwater is treated either in situ or on site (at waterworks) and land use is restricted were the most preferred. The feasibility of these alternatives is, however, questionable due to a lack of experience with large-scale applications using the suggested groundwater treatment techniques. The results are also highly dependent on site-specific data, and they are particularly sensitive to valuation of the ‘Risk reduction’ decision criterion. In practice, owing to the high costs and lack of payers, hardly any feasible RM methods suitable for Finnish shotgun shooting ranges are currently available (Sorvari et al., 2006).

4.8.4 Evaluation of the PIRTU tool

The PIRTU tool developed for site-specific evaluation of alternative RM approaches was originally based on the Dutch REC system. Several DSTs have been developed abroad for the same purpose; they are based on different calculation techniques and include a variable number of factors (decision criteria) (Table 6). Most of these tools are either commercial products, and were therefore not available for this research, or lack detailed documentation, which hindered their detailed investigation. A more detailed comparison of PIRTU and the Dutch REC system is presented in Paper III, however.

The environmental impacts included in the different DSTs vary significantly. The UvA tool seems to be the most comprehensive from this perspective, since it even includes impacts related to the production of additives used in remediation. The number of emission categories is also high, for example for adsorption and bioremediation techniques it totals almost 80. For comparison, REC only considers three types of emissions. On the other hand, the tools based solely on LCA only cover environmental effects related to remediation measures, and their usability in assessing the total merit, eco-efficiency, or sustainability of alternative RM approaches is therefore limited.

The studies using two model sites showed that, besides providing a means to systematically study the dimensions of different RM actions and identify the most eco-efficient (Sorvari et al., 2005), cost-efficient, or preferred RM option (based on group decision), the PIRTU tool is useful in promoting discussions between different stakeholders.

In practice, the uncertainties involved in the determination of the magnitude of risks, costs, and other factors, and the uncertainties of the effectiveness of RM actions have to be considered when selecting appropriate measures to eliminate/restrict risks. Moreover, the temporal dimensions of the RM alternatives are hidden in PIRTU, which needs to be taken into account when interpreting the results. Ignoring this aspect can lead to faulty conclusions, which again result in less ideal solutions, i.e. increased preference of methods that are economical but slow in practice (see Section 4.8.3 above and Paper III).

In the study by Onwubya et al. (2009), 130 experts from over 10 European countries named the following features essential in DSTs: supporting decision-making at sites with mixed contamination and different levels of contamination; integrating ecological and physicochemical traits; combining various types of soils, climates, and plants, surface and groundwater parameters, ecosystem sensitivity, and various types of remediation techniques; encouraging dialog between stakeholders and informed decision-making; being simple, pragmatic, detailed and comprehensible for practical application; and including the consideration of feasibility, costs, and application range. Incorporating the multi-criteria approach and CBA to assess socioeconomic factors was considered essential. PIRTU seems to fulfill most of these expectations, although some further development is still needed. From the modeling perspective, these development needs at least involve inclusion of a model to assess runoff that would be applicable in Finnish conditions, updating and adding life cycle data on different RM methods, and incorporating algorithms for conducting a thorough uncertainty analysis. While overall the structure of the value tree adopted in PIRTU was considered suitable, the difficulty of weighting some incomparable attributes under the ‘Environmental effects’
Table 6. Main features of some available software tools for site-specific comparison of alternative risk management actions. LCA = life cycle analysis, CA = cost analysis, CBA = cost-benefit analysis, CEA = cost-effectiveness analysis; MCDA = multi-criteria decision analysis; AHP = analytical hierarchy procedure.

<table>
<thead>
<tr>
<th>Name of the DST / Origin</th>
<th>Approach</th>
<th>Factors involved &amp; characteristics</th>
<th>Ref</th>
</tr>
</thead>
<tbody>
<tr>
<td>UvA (Umweltbilanz für Altlastensanierung) / Germany (Baden-Württemberg)</td>
<td>LCA</td>
<td>Environmental effects: 10 main impact categories, most of the secondary impacts covered</td>
<td>Anon, 1998</td>
</tr>
<tr>
<td>CARO (Cost Analysis of Remediation Options) ROCO (ROugh COst Estimation Tool)</td>
<td>CA</td>
<td>Costs; to be used for detailed cost estimation</td>
<td>SUMATECS, 2008</td>
</tr>
<tr>
<td></td>
<td>ROCO</td>
<td>Costs; to be used only for rough estimation; contamination of soil and groundwater differentiated</td>
<td>SUMATECS, 2008; WELCOME, 2004</td>
</tr>
<tr>
<td>MOKKA / Hungary</td>
<td>mass flow balance, CBA, SWOT analysis (qualitative)</td>
<td>Risks, costs, technological efficiency, and non-measurable characteristics of RM technologies</td>
<td>Gruiz et al., 2008</td>
</tr>
<tr>
<td>ABC (Assessment, Benefits and Costs) / the Netherlands</td>
<td>LCA &amp; CBA</td>
<td>Risks, costs, environmental effects; covers direct and indirect benefits (e.g. social benefits associated with risk reduction) in different spatial scales (global, regional, local)</td>
<td>Maring et al., 2003</td>
</tr>
<tr>
<td>WILMA / Germany</td>
<td>CBA</td>
<td>Economic value and ecological value (including primary and secondary effects and their weighting)</td>
<td>Weth, 2001</td>
</tr>
<tr>
<td>ROSA / the Netherlands</td>
<td>CBA modified CEA</td>
<td>Risks, costs, and reduction in liabilities</td>
<td>SUMATECS, 2008</td>
</tr>
<tr>
<td>METEORS (Model for the Evaluation of a Technically and Economically Optimal Remediation Strategy) / Canada</td>
<td>MCDA/MAVT &amp; LCA</td>
<td>Risks, costs, and environmental effects (9 main impact categories)</td>
<td>Beinat and van Drunen, 1997</td>
</tr>
<tr>
<td>REC (Risk Reduction, Environmental Merit and Costs) / the Netherlands</td>
<td>MCDA/MAVT</td>
<td>Environmental effects, costs, social factors, minimum achievable concentration, clean-up time, reliability and maintenance, residuals, site data needs, safety, post-treatment needs of by-products, applicability, acceptability, and development status of technologies</td>
<td>Vranes et al., 2001</td>
</tr>
<tr>
<td>DARTS (Decision Aid for Remediation Technology Selection) / Italy</td>
<td>MCDA/ PROMETHEE</td>
<td>Risks, costs, environmental effects, social factors (monetized); integrated with GIS; includes probabilistic risk calculation and consideration of public acceptability, efficiency, availability of the technologies</td>
<td>Agostini et al., 2009; Carlon et al., 2007</td>
</tr>
<tr>
<td>DESYRE (Decision Support System for rehabilitation of Contaminated Sites) / Italy</td>
<td>MCDA/AHP</td>
<td>Risks, costs, social factors; includes a GIS subsystem; and sensitivity, probabilistic, and fuzzy analysis</td>
<td>Sullivan et al., 2009; <a href="https://www.decerns.com">https://www.decerns.com</a> Paper III; Sorvari et al., 2005</td>
</tr>
<tr>
<td>DECERNS (Decision Evaluation in ComplEx Risk Network Systems) / Russia PIRTU (Pilaantuneen maaperän ja pohjaveden Riskinhallinnan Tukijärjestelmä) / Finland</td>
<td>MCDA/several methods, CBA, CEA MCDA/MAVT</td>
<td>Risks, costs, environmental effects, social factors</td>
<td></td>
</tr>
<tr>
<td>SMARTe (Sustainable Management Approaches and Revitalization Tools – electronic) / Germany and USA</td>
<td>RA, financial analysis</td>
<td>Risks (including fate and transport), net economic benefits; currently only separate tools, but a decision analysis tool under preparation (as of March 2009)</td>
<td><a href="http://www.smarte.org/smarthe/research/index.xml">http://www.smarte.org/smarthe/research/index.xml</a></td>
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</table>
criterion showed that it is worth studying some minor changes in the measures. In addition, work to link PIRTU with a simple RA module is ongoing.

In order to use PIRTU to identify the most sustainable RM option, the scope of the tool needs to be extended to include additional factors. Destruction of habitat associated with remediation was already considered in the model calculations under the ‘Other effects’ criterion. Additional factors relevant from the viewpoint of sustainability include mechanical, physico-chemical, and toxic impacts on soil arising from remediation. These impacts can appear as instability, changes in redox conditions, changes in the water table, and depletion of organic matter or minerals, among other things. All such additional attributes can be included in the ‘Other effects’ module. Similar qualitative assessment categories combined with their scoring used for the attributes already inbuilt within this criterion are probably required to quantify them.

5 Conclusions and recommendations

Generic benchmark values, such as soil quality guidelines, do not always provide sufficient means for managing contaminated sites, since they fail to take into account specific receptors or environmental conditions that might affect the formation and dimensions of risks. Truly risk-based land management therefore requires using more advanced techniques to assess the risks involved. The difficulty is then to choose appropriate methods and tools, since they are the key determinants of the resulting risk estimates.

There are two different approaches to risk assessment (RA) of contaminated sites. The ‘traditional’ approach relies on the precautionary principle and uses conservative methods to generate risk estimates that are in line with this principle. Whereas the alternative, more current approach, aims at applying a tiered approach and generating more realistic risk estimates in order to optimize the use of risk management (RM) resources. In such an approach, conservative methods are generally used at the lowest assessment tiers while different multimedia software tools and specific transport models are applied in higher tiers. In this research, several screening-level RA methods and multimedia tools with varying levels of detail were used to assess risks to human health and biota at some contaminated sites typical of Finland. A statistical software program was applied to assess the variability and uncertainty of risk estimates. The results showed mainly minor health risks at all the study sites even in the most sensitive, assumed future land use and therefore, no immediate RM actions or more detailed assessment were warranted. However, the dimensions of ecological risks remained unclear at most sites, indicating the need for more elaborate studies.

This research showed significant differences in the approaches, model concepts, algorithms, and parameters in the multimedia software tools used in human health risk assessment (HRA). As indicated by the previous comparison studies of multimedia tools, these disparities appear as variability in risk estimates, with differences sometimes extending over several orders of magnitude. This research also identified various shortcomings and limitations in the studied multimedia HRA tools. These shortcomings and limitations are both general and more specific, the latter referring to certain contaminant transport pathways, exposure routes, or specific contaminants. The limitations arise from the inability of simplified models to accurately account for all the factors involved in the formation of risks, as well as from the conditions in Finland, and include issues related to very lipophilic contaminants and climatic and regional conditions, to name a few. Ignoring these issues can result in erroneous or at least unrealistic risk estimates and consequently, over- or undersized RM actions. In order to be able to prioritize one method over another, one needs to know its principles, usability, limitations, and sensitivity (to input data). Even though comparison studies of different tools for HRA have undoubtedly provided useful information on the disparities of different tools, such studies also involve a risk of creating mistrust towards
using them among decision-makers and other stakeholders, particularly if the differences cannot be clearly and understandably justified and the results of the studies do not allow appointing ‘the right tools to use’. Therefore, rather than conducting additional comparison studies, more effort should be directed to validating the key algorithms in RA with true monitoring data in order to define the most suitable models for different cases in Finland (considering contaminants involved, environmental conditions, and exposure scenarios). Such validation would cover, for example, plant uptake and vapor intrusion models.

Applying sophisticated and data-intensive methods right from the beginning of the assessment of human health risks caused by land contamination has so far been common practice in Finland. This research does not justify such an approach since in many cases it turned out that even simple, conservative RA methods can provide adequate information for decision-making concerning RM actions, thereby making the use of more laborious and complex methods unnecessary. In practice, the availability of time, money, and expertise should be considered in the selection of RA methods, since these factors affect the quality and contents of RA. A tiered approach proceeding from simple methods to more complicated ones then provides the most economical and a scientifically sound way to conduct RA and has in fact been adopted in RA frameworks in several countries. It is important to further promote broad adoption of this approach in Finland.

Compared to HRA, ecological risk assessment (ERA) involves more uncertainties due to the diversity of organisms. In ERA, the inability of simple screening-level methods based on exposure and uptake models or risk-based benchmark values to account for the complexity of ecosystems, significantly limits the usefulness of the results of such assessments in decision-making. Screening-level methods often produce overly conservative risk estimates and consequently, indicate a need for extensive RM actions that are practically non-feasible. Moreover, the ecological benchmarks used in lower assessment tiers to determine risks vary considerably, causing high uncertainty in the results. Also, there is insufficient information on the suitability of different models (and the parameters involved) in Finnish conditions. Chemical studies alone are seldom sufficient to determine risks to biota, since they fail to produce sufficient information on the combined effects of several contaminants and phenomena such as adaptation and avoidance, among others. The need for some biological studies to verify the magnitude and scale of ecological risks was actually identified in most of the case studies included in this research. Ecotoxicological tests and some biomonitoring are thus highly recommended to be carried out alongside and partly in place of chemical studies.

The TRIAD procedure, based on using three alternative approaches, i.e. chemistry, ecotoxicity, and ecology, has recently been suggested as a solution for increasing the reliability of ERA. While this research proved the suitability of TRIAD in the assessment of a site contaminated by multiple chemicals, the limitations of the procedure, and potential problems that emerged, need to be acknowledged. Biological methods can produce data which are not readily suitable for processing with the straightforward mathematical methods applied in TRIAD. For example, hormesis and a lack of proper reference soil can limit the usability of the results. The latter problem calls for additional sampling and studies, for which resources should be available. Even then, in practice it can turn out to be impossible to find a true reference site or reference soil to compare with. The results from different methods can also produce controversial results, necessitating more detailed analysis. TRIAD is in fact a tiered procedure comparable to the tiered approach adopted in HRA. The lowest tier in TRIAD-based ERA should therefore involve simple, cost-effective methods. It is advisable to adopt TRIAD already in the planning stage of ERA, keeping in mind the study problem (e.g. characteristics of the site, contaminants, and potential receptors involved) and the tiered approach, since this allows optimization of the methods.

ERA practices are still only developing in Finland, as in many other countries. While this research mainly focused on using simple screening-level ERA tools, which fail to reliably assess risks on the population scale, at large contaminated areas, other tools that address effects on a wider
scale, such as ecosystem-level models (described, e.g. in Preziosi and Pastorok, 2008) should be adopted. So far such tools have not been applied in site-specific CLM in Finland. Finally, the key questions related to ERA still remain, i.e. what do we want to protect at a particular site, and what is the desired level of protection. These are merely policy questions that should be solved by risk managers (and policy-makers) prior to even planning the ERA. This discussion is, however, only beginning in Finland.

Besides the suitability of the models and multimedia HRA tools, the lack of representative parameter data is a key issue and can limit the use of very detailed models and tools in practice. Although there are databases available that include exposure parameters, these mainly comprise American and European data. While some of these data are generic and applicable irrespective of the country, verification of the applicability of some parameters in Finnish conditions is needed. On the other hand, using foreign data can be justified in producing screening-level risk estimates in the lower RA tiers. Variation in benchmarks (ERA) and reference values (HRA) used to determine risk estimates is a problem that has received little attention in Finland, nor has it been adequately brought up in the previous RAs. This variation can result in significant differences in risk estimates when different tools and databases are used. It is therefore advisable to study the basis of the benchmarks and reference values included in the tools and available in the literature and other sources in order to verify their appropriateness in Finland and in the specific case under study. While some data are already available in different literature sources and databases, all these scattered information should be analyzed and the validated data compiled into a database that could be utilized in future RAs in Finland.

Statistical methods such as the Monte Carlo simulation can provide useful information on the uncertainty and variability of risk estimates. Their use requires data on the statistics of the key variables involved as well as on correlations between them. Correlations have generally been ignored in statistical analysis and due to a lack of data, they were not considered in this research, either. Determining correlations between key variables therefore calls for additional studies. The accuracy of statistical numbers, particularly the form of distributions, should also be studied in more detail if statistical tools are used. While Monte Carlo simulation and other equivalent statistical methods do not provide a means to systematically consider the problems or limitations related to the methods per se, they can be studied using the multi-criteria analysis (MCA) techniques. This research applied MCA by ranking the separate ERA methods on the basis of their performance in terms of predicting risks at the study site. The results proved that such a process allows consideration of factors that reduce the reliability of the results, such as sample pretreatment and relevance of the test organisms or the measurement endpoints from the viewpoint of the study site. This undoubtedly increases the reliability of risk estimates.

It is noteworthy that risks are only one of the multiple factors involved in decision-making concerning contaminated sites, and in some cases they are not even the most important driver in the selection of RM actions. Directing all efforts to generating very accurate risk estimates is not meaningful if other factors not strictly related to the magnitude and scale of the risks, such as maintaining a positive public image, are actually driving the selection of RM actions.

In fact, achieving sustainability or eco-efficiency in CLM requires consideration of additional factors besides risks; the importance of these other contributing factors (i.e. decision criteria) varies case by case. This research involved developing a calculation tool known as 'PIRTU' for such a purpose, using the Dutch REC system as a starting point. PIRTU proved to be useful in identifying the optimal or preferred remediation option when multiple decision criteria are involved. It also provides a means for discussions and communication between different stakeholders, which has been considered one of the desirable features of decision support tools (DST). It is worth noting that PIRTU is primarily intended for cases where no best RM alternative is easily identifiable without a multifaceted analysis. Moreover, planning and conducting the weighting procedure in
particular, requires adequate expertise in order to produce reliable results. Further development of this DST should involve incorporating a Finnish risk assessment tool (ongoing); including algorithms for assessing runoff; attaching a module for considering the time span of RM actions; and adding more life cycle data on alternative remediation techniques. In addition, inclusion of some additional sustainability factors and linking with an uncertainty analysis tool would increase the usability of PIRTU. Such uncertainty analysis should extend beyond all decision criteria, attributes, and sub-attributes involved.

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