



A decision support tool to prioritize risk management options for contaminated sites

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ABSTRACT

The decisions on risk management (RM) of contaminated sites in Finland have typically been driven by practical factors such as time and money. However, RM is a multifaceted task that generally involves several additional determinants, e.g. performance and environmental effects of remediation methods, psychological and social factors. Therefore, we adopted a multi-criteria decision analysis approach and developed a decision support tool (DST) that is viable in decision-making in such a complex situation. The basic components of the DST are based on the Dutch REC system. However, our DST is more case-specific and allows the consideration of the type, magnitude and scale of contamination, land use, environmental conditions and socio-cultural aspects (e.g. loss of cultural heritage, image aspects). The construction of the DST was started by structuring the decision problem using a value tree. Based on this work we adopted the Multi-Attribute Value Theory (MAVT) for data aggregation. The final DST was demonstrated by two model sites for which the RM alternatives and site-specific data were created on the basis of factual remediation projects and by interviewing experts. The demonstration of the DST was carried out in a workshop where representatives of different stakeholders were requested to rank and weight the decision criteria involved. To get information on the consistency of the ranking of the RM alternatives, we used different weighting techniques (ratio estimation and pair-wise weighting) and alternative ways to treat individual respondents' weights in calculating the preference scores for each RM alternative. These dissimilar approaches resulted in some differences in the preference order of the RM alternatives. The demonstration showed that attention has to be paid to the proper description of the site, the principles of the procedure and the decision criteria. Nevertheless, the procedure proved to enable efficient communication between different stakeholders and the identification of the preferred RM option.

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1. Introduction

Decisions regarding the risk management (RM) of contaminated sites in Finland have typically been based on generic guideline values which do not consider site-specific risks (Sorvari and Assmuth, 2000; Mäenpää, 2002). In most cases, the direct costs, time and achievable risk reduction are still the only criteria involved in the decision-making (Sorvari, 2005; Sorvari et al., 2009). Hence, other factors, such as overall environmental effects and social impacts, have generally been ignored or at least they have not been systematically assessed.

Soil excavation and replacement with clean soil is still the most common remediation method in Finland (Pajukallio, 2006). Excavated soil, either treated or untreated, is considered waste and it is mainly disposed of or reused in different structures and for daily cover in landfills, while recycling elsewhere is minimal (Jaakkonen, 2008). The sustainability of soil replacement and remediation based on generic guideline values, which are not strictly risk-based, has been questioned. Moreover, groundwater is usually treated with pump-and-

treat methods which has often proved to be uneconomical, time-consuming and hence, non-eco-efficient (Sorvari et al., 2009). At the end of 2003 we launched the project 'Eco-efficient risk management of contaminated soil and groundwater' to study these problems. The main goal of the project was to promote the realization of eco-efficiency in contaminated land management (CLM). Albeit eco-efficiency was previously studied in different contexts and in various industries in Finland (e.g. Seppälä et al., 2002; Melanen et al., 2004), this project is the first attempt to study it systematically in the context of CLM.

In the first phase of the project, we defined what eco-efficiency means in the context of CLM. According to a narrow definition, eco-efficiency can be described as the ratio of ecological to economic factors or vice versa (e.g. OECD, 1998; EEA, European Environment Agency, 2001) whereas a broader definition also covers social aspects i.e. human welfare (e.g. WBSCD, World Business Council for Sustainable Development, 2009). Within our project we adopted the latter approach (Sorvari et al., 2009). It turned out that in Finland, the lack of established assessment methods and guidelines is one of the main barriers to the realization of eco-efficiency in CLM (Sorvari, 2005; Sorvari et al., 2009). Stakeholder participation is also regarded as important in the attainment of eco-efficient and acceptable RM solutions.

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Several systems and techniques exist to facilitate decision-making when processing and aggregating multidimensional information and stakeholder involvement are needed. These techniques were applied in decision support tools (DSTs¹) developed for various purposes in environmental protection, including CLM. The major advantages of using such DSTs arise from the robustness, consistency, transparency and reproducibility of the decision-making process (e.g. Sullivan et al., 2001; EuroDemo, 2005).

The multi-criteria decision analysis (MCDA) is a technique commonly applied in the DSTs that handle multidimensional data. MCDA covers a group of methods by which a formal or an informal structure can be applied to the treatment of multi-objective or multi-criteria decision-making problems (e.g. Keeney, 1992; Chen et al., 1992). While MCDA approach has been used in CLM in other countries (e.g. Bonano et al., 2000; Weth, 2001; Linkov et al., 2004; EuroDemo, 2005; Kiker et al., 2005; Harbottle et al., 2006; Critto et al., 2006; Agostini et al., 2009), there are only two published cases of using it in Finland. These dealt with choosing remediation methods for a broad, multi-contaminated industrial site in the capital city area (Hokkanen et al., 2000) and for a former industrial landfill (Lahdelma et al., 2001). It is noteworthy that also at the European level the use of DSTs in CLM is still marginal (EuroDemo, 2005).

Some of the existing DSTs focus on e.g. site characterization and/or planning of sampling strategy rather than on the selection of remediation technologies. The DSTs designed for selecting remediation methods include the Dutch REC system and ABC (Assessment, Benefit, Cost) tool; the German WILMA; the Italian DESYRE (DEcision Support sYstem for REqualification of contaminated sites) and DARTS (Decision Aid for Remediation Technology Selection); DECERNS (Decision Evaluation in Complex Risk Network Systems); and the free, internet based SMARTe. The complexity, inputs and outputs as well as the bases and methods involved in these DSTs vary; nevertheless, they are all founded on the principles of life cycle analysis (LCA). However, different system boundaries and environmental impact categories, among others, can result in differing LCA results (e.g. Anderson, 2003).

From the abovementioned DSTs, the ABC tool (Maring et al., 2003) and WILMA (Weth, 2001) are both based on cost-benefit analysis. The ABC tool covers different spatial scales (global, regional, local) of both direct and indirect benefits (Maring et al., 2003). WILMA (Weth, 2001), ABC (Maring et al., 2003) and REC (Beinat and van Drunen, 1997) deliver the results classed under the separate decision criteria i.e. the results are not aggregated. DECERNS is a single software package where the tools for human and ecological risk assessment, decision analysis, economic analysis and incorporating social choices, are integrated (Sullivan et al., 2009). DECERNS includes several MCDA tools and tools to conduct cost-benefit analysis or cost-effectiveness analysis. At present, SMARTe only comprises analysis tools for considering the different aspects of CLM while the decision analysis tool is under preparation (SMARTe, 2009). The remaining DSTs use MCDA techniques with different decision criteria prioritization methods, such as the PROMETHEE outranking technique (DARTS) (Vranes et al., 2001) and the analytic hierarchy process (AHP) (DESYRE) (Carlon et al., 2007). Some DSTs e.g. DESYRE and DECERNS also combine spatial analysis, i.e. Geographical Information System (GIS), and statistical methods with the MCDA techniques.

There are no generally approved methods in Finland to systematically study the various factors involved in the decision-making on CLM and therefore, developing a flexible system – a DST – that would be suitable for evaluating the different consequences associated with the risk management of Finnish contaminated sites became the main

objective of our study. Such DST would consider the quality and dimensions of the contaminated sites, life cycle data and the prevailing environmental conditions in Finland. The DST would enable the identification of the best, i.e. the most eco-efficient/sustainable, RM option. This paper summarizes the characteristics and principles of our DST and presents an overview of its interactive demonstration with model sites and the decision-making process involved. Finally, we critically evaluate the DST and identify some further development needs.

2. Material and methods

We used the Dutch REC² system as a starting point for developing our DST mainly due to its availability and transparency. However, several modifications had to be made to make the DST more suitable for our purpose.

2.1. MCDA technique

We chose the Multi-Attribute Value Theory (MAVT) as the theoretical basis of our DST. There were two reasons for this. Firstly, MAVT is one of the major decision theories for the multi-criteria decision analysis with well established theoretical foundations (von Winterfeldt and Edwards, 1986). It can be considered a theory for the value measurement in which there are no uncertainties about the consequences of the alternatives in a decision problem. Secondly, it appeared that the REC system and the calculation rule typically used to aggregate environmental impacts in life cycle based approaches directly correspond to MAVT (Beinat and van Drunen, 1997; Seppälä, 1999; Finnveden et al., 2002). Therefore, the identical theoretical basis allowed constructing a theoretically consistent system.

The first phase of MAVT includes the structuring of the decision problem using a value tree. In the construction and definition of the elements of the value tree we considered the properties generally required, i.e. completeness, operationality, decomposability, absence of redundancy and minimum size (see Keeney and Raiffa, 1976; von Winterfeldt and Edwards, 1986). Our value tree includes the alternative site-specific RM approaches and four factors generally involved in RM decisions, known as decision criteria. These criteria are: the achievable risk reduction, costs, environmental effects and other factors. The latter criterion includes social factors and adverse effects on ecosystems and landscape associated with invasive remediation techniques. The criteria are further divided into several sub-criteria called attributes. Furthermore, the attributes are divided into sub-attributes (Fig. 1). The value of each attribute and sub-attribute defines the total value of each criterion that is, the degree to which each objective is achieved.

In the MAVT approach, the attractiveness of each RM alternative (a_j) ($j = 1, \dots, m$) is defined on the basis of criteria X_c ($c = 1, \dots, 4$). The measurement level of criterion X_c is expressed by value scores x_c . Thus, consequences $x_1(a_j) \dots x_4(a_j)$ of criteria are associated with each alternative a_j . Each criterion can be handled separately and the preference order of the RM alternatives within each criterion can then be calculated as per the following additive value function (von Winterfeldt and Edwards, 1986):

$$V_c(a_j) = \sum_{i=1}^n w_{c,i} \cdot v_{c,i}(x_{c,i}(a_j)), \quad j=1, \dots, m \quad (1)$$

where $V_c(a_j)$ is the value score, i.e. preference score, of criterion X_c ($c = 1, \dots, 4$) for RM alternative a_j , $v_{c,i}(\cdot)$ is the value function of single attribute $X_{c,i}$, and $w_{c,i}$ is the weight of that attribute within criterion X_c . The higher the $V_c(a_j)$, the more desirable the particular RM alternative is in

¹ According to Bardos et al. (2003) DSTs are “documents or software produced with the aim of supporting decision-making, i.e., something that carries out a process in decision-making”. However, here we have adopted a narrower definition and restrict the DSTs to quantitative multi-criteria models while e.g. qualitative guiding documents are explicitly excluded.

² REC comes from the Risk reduction (R), Environmental merit (E), Costs (C) (Beinat and van Drunen, 1997; van Drunen et al., 2005).

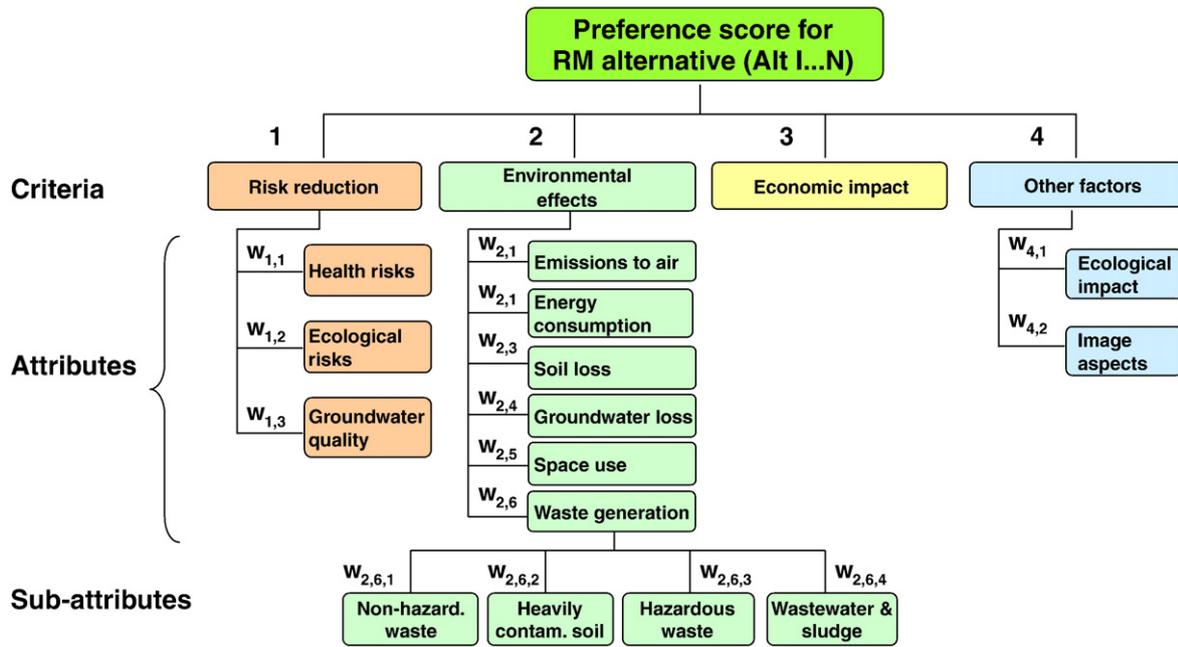


Fig. 1. Decision criteria, attributes and sub-attributes included in the decision support tool and the hierarchy between them. Only those factors involved in the two model sites are shown. RM = risk management, w = weight.

terms of criterion X_c . The shape of the value function of attribute $X_{c,i}$ can be linear or non-linear depending on the decision-makers' preferences related to the values of attribute $X_{c,i}$. This also applies to any sub-attributes.

In an additive value function, the values of $w_{c,i}$ should indicate the relative importance of the change of each attribute from its least desirable to its most desirable level (von Winterfeldt and Edwards, 1986). Before calculating the preference scores, the sum of the weights has to be normalized to 1 (Eq. (2)).

$$\sum_i w_{c,i} = 1, \quad c = 1, \dots, 4 \tag{2}$$

In our DST, we assumed linear value functions in order to arrive at a simple model. In addition, we normalized the values of each attribute function between the values 0 and 100 as is customary in MAVT. Then, if attribute $X_{c,i}$ is not divided into sub-attributes, the value function elements $v_{c,i}(X_{c,i}(a_j))$ in Eq. (1) are defined using Eq (3) (see e.g., von Winterfeldt and Edwards, 1986).

$$v_{c,i}(X_{c,i}(a_j)) = \frac{X_{c,i}(a_j) - X_{c,i}^0}{X_{c,i}^* - X_{c,i}^0}, \quad c = 1, \dots, 4 \tag{3}$$

where $X_{c,i}^0$ is the lowest and $X_{c,i}^*$ is the highest score of attribute $X_{c,i}$.

If attribute $X_{c,i}$ is divided into sub-attributes $X_{c,i,l}$, the value function elements $v_{c,i}(X_{c,i}(a_j))$ in Eq. (1) are determined on the basis of Eq. (4) (see e.g., von Winterfeldt and Edwards, 1986).

$$v_{c,i}(X_{c,i}(a_j)) = \sum_{l=1}^r w_{c,i,l} \cdot \frac{X_{c,i,l}(a_j) - X_{c,i,l}^0}{X_{c,i,l}^* - X_{c,i,l}^0}, \quad c = 1, \dots, 4; i = 1, \dots, n \tag{4}$$

where $X_{c,i,l}(a_j)$ is the value score of alternative a_j for sub-attribute $X_{c,i,l}$, $w_{p,i,l}$ is the weight of that sub-attribute, and $X_{c,i,l}^0$ is the lowest and $X_{c,i,l}^*$ is the highest score for that sub-attribute. According to MAVT, the values of $w_{c,i,l}$ should indicate the relative importance of changing each sub-attribute from its least desirable to its most desirable level and the sum of $w_{c,i,l}$ should equal 1.

Finally, we can calculate the total preference score for each RM alternative by combining the attribute values for each decision criterion (Eq. (5)).

$$V(a_j) = \sum_{c=1}^4 p_c \cdot V_c(a_j), \quad j = 1, \dots, n \tag{5}$$

where $V(a_j)$ is the total preference score for RM alternative a_j , p_c is the weight of criterion c and $V_c(a_j)$ is the preference score of criterion X_c for RM alternative a_j . Again, the values of p_c should indicate the relative importance of changing each criterion from its least desirable to its most desirable level and the sum of p_c should equal 1. The preference of each RM alternative is shown by a total preference score meaning that the higher the score the better the alternative (= higher preference). It is notable that the calculation rules of the above-mentioned preference model are assumed to fulfill the assumption concerning the difference independence between attributes of each criterion. This assumption is necessary when using the additive model. The validity of the assumption was tested by asking the participants of the weighting task if they can think of preferences for several levels of attributes independently from the levels of other attributes. All participants stated that they can.

2.2. Model sites and their risk management alternatives

The model sites created for testing and elucidating our DST included an outdoor shotgun shooting range and a former gasoline station (Table 1). These represent common types of contaminated

Table 1 Description of the model sites studied.

	Size, m ²	Contaminants	Location	Land use scenario	GW involved
Shooting range	160,000	Pb (As, Sb)	Rural	GW uptake, recreation (as it stands)	Yes
Gasoline station	15,000	PHCs	Urban	Housing, no GW uptake	Yes

GW = groundwater, PHCs = petroleum hydrocarbons.

sites in Finland but are very different from the risk management perspective.

Former gasoline stations comprise about one third of all registered contaminated or potentially contaminated sites in Finland (Finnish Environment Institute, 2009). They can generally be characterized by the following features: small area, contaminated groundwater (or serious risk of groundwater contamination) and availability of feasible soil remediation methods. While shotgun shooting ranges typically cover several hectares but less frequently, pose a serious threat to groundwater quality. Moreover, presently there are hardly any economically feasible methods to remediate them. According to the national survey, the number of shooting ranges in Finland totals 2000–2500 (Sorvari et al., 2006), that is some 10% of all contaminated or potentially contaminated sites.

For the model sites, we defined several risk management scenarios (i.e. RM alternatives) including 'traditional' ex situ and more novel on site and in situ remediation techniques (Table 2). The definition of the RM alternatives was based on the knowledge of the most common remediation methods used at present and the most relevant new technologies. This information was collected from several previous case documents (unpublished reports) and by interviewing some Finnish CLM experts.

2.3. Determination of value scores for decision criteria

To determine value scores $x_{c,i}(a_j)$ and $x_{c,i,l}(a_j)$ for attributes $X_{c,i}$ and sub-attributes $X_{c,i,l}$ associated with different RM alternatives a_j , we created site-specific data on the basis of factual remediation projects and by interviewing several experts representing service providers. Temporal boundaries varying from 20 to 30 years were used in previous studies on the life cycle extending consequences of site remediation (e.g. Beinart and van Drunen, 1997; Diamond et al., 1999; EuroDemo, 2007). In compliance with these studies, we adopted the time span of 30 years in our DST.

Based on the site-specific data we calculated risk indexes for the attribute 'Health risks' under the criterion 'Risk reduction' using the Risc-Human software version 3.1. (by van Hall Instituut). The results were given as input to the DST. The risk indexes associated with other risks were determined as a ratio of the environmental concentration to a suitable benchmark for that particular medium, such as the target concentration for soil or quality standard for domestic water (Table 3). Under the criterion 'Other factors', values were defined by expert judgments based on a qualitative scale. Whereas the scores for the attributes 'Emissions to air' and 'Energy consumption' under the criterion 'Environmental effects' were determined on the basis of the Finnish life cycle data and using methods of the Finnish LIPASTO calculation system (available at: <http://lipasto.vtt.fi/indexe.htm>) and REC. Lastly, data on the costs of different remediation methods was obtained from the contractors, treatment plants and developers of remediation technologies. The final values of the attributes and sub-attributes associated with each RM alternative are presented in Table 4.

2.4. Determination of weights

After the definition of numeric values for all attributes and sub-attributes involved in the decision-making, attribute and criteria weights need to be set. For this purpose, we prepared forms and background material that described the study method as well as the model sites and their RM alternatives, and tested these with a few CLM experts from the Finnish Environment Institute (SYKE). The experimenters' comments and possible problems that arose during the weighting process were registered and the material was revised accordingly. At the next stage, we organized a stakeholder seminar for invited experts to whom we sent the modified background material. In the seminar we again introduced and discussed the study problems and the DST and asked the participants to value the criteria, attributes and sub-attributes involved in the model sites by giving

Table 2

Risk management (RM) alternatives for the model sites. GW = groundwater; BTEX = benzene, toluene, ethylbenzene and xylenes; TVOC = total volatile organic compounds; PHC = petroleum hydrocarbons; SVE = soil vapor extraction; MNA = monitored natural attenuation; na = not available.

RM alternative	Method	Remedial targets	Volume of soil and GW treated (m ³)
A. Shooting range			
Alt 0	No soil remediation; closure of water intake, building of a new waterworks	–	–
Alt I	Soil excavation + landfill disposal; closure of water intake, building of a new waterworks	Soil guideline values (old): As 10 mg kg ⁻¹ ; Pb 60 mg kg ⁻¹ ; Sb 5 mg kg ⁻¹	Soil: 45,000
Alt II	Soil excavation + landfill disposal; closure of water intake, building of a new waterworks	Upper soil guideline values (new): As 160 mg kg ⁻¹ ; Pb 520 mg kg ⁻¹ ; Sb na	Soil: 16,500
Alt III	Soil washing + reuse on site; closure of water intake, building of a new waterworks	See Alt I	Soil: 45,000
Alt IV	Top soil (0.01 m) including the shots excavated, shots recycled + land use restricted; GW treated in situ by a reactive barrier	No target for soil, estimated Pb removal 70%; GW below the quality standards for domestic water: 10 µgPb l ⁻¹	Soil: 1,300
Alt V	No soil remediation, land use restricted; GW treated on site (at waterworks) by Metclean technique	No target for soil; GW: See Alt IV	–
Alt VI	No soil remediation, land use restricted; GW treated on site (at waterworks) by membrane filtration	See Alt V	–
B. Gasoline station			
Alt 0	No remediation	–	–
Alt I	Soil excavation +	Soil limit values (old): xylenes 25 mg kg ⁻¹ ; TVOC 500 mg kg ⁻¹ ; fuel oil, light 1000 mg kg ⁻¹ GW: BTEX <10 µg l ⁻¹ , TVOC 1000 µg l ⁻¹ , heavier PHCs 1000 µg l ⁻¹ ,	Soil: 805 (V) GW: ca. 500
	a. soil composting and reuse on site		
	b. landfill disposal		
	c. combustion off site		
	GW treated in situ by absorption to activated carbon		
Alt II (a, b, c)	See Alt. I (a, b, c)	Soil guideline values (old): xylenes 0.5 mg kg ⁻¹ ; TVOC 100 mg kg ⁻¹ ; fuel oil, light 300 mg kg ⁻¹ , GW: see Alt. I	Soil: 1,978 (V) GW: see Alt. I
Alt III	MNA	Final concentrations in soil and GW defined on the basis of the data from a Finnish research project	–
Alt IV	SVE (6 months) + MNA	See Alt III	Soil: 0 GW: 360

Table 3
Description of the methods used in the definition of values for the attributes and sub-attributes involved in the model sites.

Criterion/attribute	Determination of attribute value
<i>Risk reduction</i>	
Health risks	Risk reduction ^a , $RR_h[\%] = 100 * (r_{h, tot, present} - r_{h, tot, before/during/after\ remediation}) * (r_{h, tot, present})^{-1}$ where $r_{h, tot} [-]$ is a health risk estimate that considers the magnitude and scale of risks (number of people that are potentially exposed); $r_{h, tot}$ is calculated from risk estimates representing different phases of remediation (r_h values) by: $r_h = A * N * t_{phase} * RL_h$, and $r_{h, tot} = \sum r_h / 30\ a$, where A is the area of the site [m^2], N is the number of receptors (people) per area [m^2] (depends on the land use), t_{phase} is the duration of the remediation phase (before, during or after remediation) [a], RL_h (dimensionless) is the risk index implying health risks (value to be calculated using a separate software).
Ecological risks, terrestrial ecosystem	Risk reduction ^a , $RR_e [\%] = 100 * (V_{soil, present} - V_{soil, before/during/after\ remediation}) * (V_{soil, present})^{-1}$ Risk expressed as the volume (V_{soil}) of contaminated soil (proportioned to soil reference value that is based on ecological risks), $V_{soil} [eq - m^3] = \sum m_x / (\rho * C_{X,T})$; $m_x = \rho * (C_x - C_{X,T}) * A_x * h_x$, where m_x [mg] is the average soil load related to contaminant X during 30 years, ρ is the bulk density of soil [$kg\ m^{-3}$], C_x is the concentration of contaminant X in soil [$mg\ kg^{-1}$], $C_{X,T}$ [$mg\ kg^{-1}$] is the soil reference value based on ecological risks i.e. the Finnish target concentration of contaminant X in soil, A_x is the size of the area [m^2] contaminated by contaminant X , and h_x is the depth of soil layer [m] contaminated by contaminant X .
Groundwater quality	Risk reduction ^a , $RR_{gw} [\%] = 100 * (L_{gw, present} - L_{gw, before/during/after\ remediation}) * (L_{gw, present})^{-1}$ Risk expressed as groundwater load (L_{gw}) that considers the contamination level and the toxicity of the separate contaminants involved, $L_{gw} [eq - \mu g\ l^{-1}] = \sum C_x * ef_x$, where C_x is the concentration of contaminant X in the saturated zone [$\mu g\ l^{-1}$] and ef_x is the equivalence factor of contaminant X describing the toxicity of that contaminant in relation to other contaminants.
<i>Environmental effects</i>	
Soil loss	Use of soil, $Loss_{soil} [m^3] = \text{clean soil transported to the site } [m^3] - \text{excavated soil reused on/off site}$
Groundwater loss	Groundwater lost due to contamination, $Loss_{gw} [m^3] = \text{volume of removed groundwater } [m^3] - \text{volume of groundwater recycled into soil } [m^3]$
Energy consumption	Consumption of diesel, oil, gas, electricity, EC [inhabitant-eq] = (energy used in soil treatment [MJ] + energy used in excavation [MJ] + energy used in transportation [MJ]) / annual energy consumption per inhabitant in Finland [MJ] where separate energy consumptions are calculated by Treatment of 1 ton of soil: nominal output [kW] * specific energy consumption [MJ/kWh] / treatment efficiency [t/h] Excavation: amount of soil excavated [t] * energy consumption [MJ/t] Transportation: Amount of soil transported [t] * distance [km] * fuel consumption [MJ/tkm]
Emissions to air	Air emission index [Finnish inhabitant-eq] The calculation is based on life cycle impact assessment methodology. Emissions of CH_4 , CO_2 , SO_2 , PM, VOC, N_2O , and NO_x are multiplied by characterization factors for climate change, acidification, ozone formation and eutrophication (see Seppälä et al., 2006). The calculated indicator results are divided by the indicator values of the Finnish economy. The normalized results are multiplied by impact category weights (Seppälä, 1999) in order to aggregate the indicator results into one score. Finally, the total score is divided by the number of inhabitants in Finland. The emissions are calculated in the following way: Transportation: emissions [kg] = Amount of soil transported [t] * distance [km] * emissions per distance [g/tkm] / 1000 Excavation: emissions [kg] = Amount of soil excavated [t] / (capacity [t/h] * nominal output kW * specific emission [g/kWh])
Waste generation	Volume [m^3] of – Non-hazardous waste – Heavily contaminated soil – Hazardous waste – wastewater and sludge To be assessed, depends on the remediation method.
Space use	Area [m^2] which is non-usable due to contamination or ongoing remediation activities * duration of the phase [a]
<i>Other factors</i>	
Ecological impact ^b	Impact index [dimensionless] = magnitude of impact [-] * number of ecological receptors per area [number m^{-2}] * size of the area [m^2]; the magnitude of impact is defined by expert judgment using a qualitative scale that is quantized [dimensionless]. Number of ecological receptors depends on land use. Scale: significant positive impact (+3) – moderate positive impact (+2) – minor positive impact (+1) – no impact (0) – minor negative impact (-1) – moderate negative impact (-2) – significant negative impact (-3).
Image aspects	Impact index is defined using a quantitative scale determined by expert judgment [dimensionless]. Scale: significant positive impact (+3) – moderate positive impact (+2) – minor positive impact (+1) – no impact (0) – minor negative impact (-1) – moderate negative impact (-2) – significant negative impact (-3).

^a If there are no remedial actions the risk estimates referring to the contaminated soil during and after remediation receive the value of the current situation (if the land use remains unchanged) whereas in the case of groundwater these risk estimates differ when some natural attenuation of contaminants is expected to occur. In the case of remediation, the values with the footnote "before remediation" refer to the risks prior to remediation activities are in This refers to adverse effects to biota caused by remediation.

^b This refers to adverse effects to biota caused by remediation.

them weights. The experts involved in this weighting process comprised service providers (5), regional and municipal environmental authorities (4), problem owners (3), a representative from the Ministry of the Environment and researchers and experts from SYKE (6) and from other public institutes (3) representing different CLM expertise. To complement the material, six permitting authorities from different regional environment centers carried out the valuation task in connection with the national CLM seminar. Unfortunately, due to time constraints we were able to conduct the valuation only for the gasoline station.

We used the weighting based on the ratio estimation technique (von Winterfeldt and Edwards, 1986). Weights were defined starting from the sub-attributes. In each group of factors, i.e. sub-attributes, attributes and criteria, the attendees were first advised to rank the factors starting from the most important and ending up to the least important. Then, they should address a value of 10 to the sub-attribute/attribute/criterion which they had ranked as the lowest, i.e. the least important in their decision-making, while a value of > 10 should be addressed to other sub-attributes/attributes/criteria as per their relative importance compared with the least important factor in

Table 4

Attribute, sub-attribute and ‘Costs’ criterion values for the shooting range (A) and gasoline station (B). See Fig.1 for the hierarchy between the criteria, attributes and sub-attributes.

Criterion, attribute, sub-attribute	Alt 0	Alt I	Alt II	Alt III	Alt IV	Alt V	Alt VI		
A. Shooting range									
Risk reduction (%)									
Health risks ^a	0	94	74	94	84	84	84		
Ecological risks	0	99	87	99	70	0	0		
Groundwater quality	0	0	0	0	33	33	33		
Environmental effects									
Soil loss (m ³)	0	45,000	11,000	0	0	0	0		
Energy consumption (inhabitant-eq)	0	32	10	29	1.3	1.0	1.0		
Emissions to air (inhabitant-eq)	0	123	40	71	18	2.3	2.3		
Waste generation (m ³)									
– Heavily contaminated soil	0	33,000	7,500	0	0	3,000	12,000		
– Hazardous waste	0	12,000	9,000	0	0	0	0		
– Wastewater and sludge	0	0	0	2,000	0	0	0		
Space use (m ² year)	0	210,000	155,000	93,000	4800,000	4800,000	4800,000		
Costs (k€)	1,475	5475	2,646	4,044	777	347	514		
Other factors									
Ecological impact	0	–7,800	–2,400	–7,800	–5,200	0	0		
Image aspects	–1,600	2,400	800	2,400	800	–2,400	–2,400		
Criterion, attribute, sub-attribute	Alt 0	Alt Ia	Alt Ib	Alt Ic	Alt IIa	Alt IIb	Alt IIc	Alt III	Alt IV
B. Gasoline station									
Risk reduction (%)									
Health risks ^a	0	31	31	31	90	94	94	61	77
Ecological risks	0	68	68	68	95	95	95	71	75
Groundwater quality	0	97	97	97	97	97	97	96	95
Environmental effects									
Soil loss (m ³)	0	0	805	805	0	1,978	1,978	0	0
Groundwater loss (m ³)	0	0.5	0.5	0.5	0.5	0.5	0.5	0	0.36
Energy consumption (inhabitant-eq)	0	0.12	0.70	22	0.20	1.6	55	0	1.9
Emissions to air (inhabitant-eq)	0	0.19	2.7	73	0.90	6.3	178	0	4.4
Waste generation (m ³)									
– Heavily contaminated soil	0	0	805	805	0	1,978	1,978	0	0
– Hazardous waste	0	55	55	55	55	55	55	0	55
– Wastewater and sludge									
Space use (m ² year)	450,000	12,500	15,000	15,000	17,500	18,750	18,750	450,000	247,500
Costs (k€)	6.9	127	163	191	240	327	400	196	166
Other factors									
Ecological impact	0	–0.0014	–0.0014	–0.0014	–0.0034	–0.0034	–0.0034	0	0
Image aspects	–75	75	150	150	150	225	225	75	150

^a In the software tool used for the assessment of health risks, the TDI (tolerable daily intake) value used in the characterization of the risks covers both carcinogenic and non-carcinogenic effects.

that particular group of sub-attributes/attributes/ criteria. For example, if the ‘Emissions to air’ is regarded as the least important attribute under the criterion ‘Environmental effects’, a value of 10 should be addressed to this attribute. Then, if the attribute ‘Energy consumption’ is considered twice as important, this attribute should receive a value of 20. A value of 0 should be given to all those attributes (and sub-attributes and criteria) that are found totally indifferent in decision-making.

To study the effect of weighting method, we also carried out a pair-wise weighting (Saaty, 1980) of the four criteria. This study was only executed for the gasoline station and due to time constraints, only six experts participating in our seminar carried out the weighting using the two methods. In pair-wise weighting, each single criterion is compared with another criterion and hence, in the case of four decision criteria there are six pairs (= (n – 1)!) to compare. Weighting was conducted individually with each person using the Hipre software developed in the Helsinki University of Technology, System Analysis Laboratory (available at www.hipre.hut.fi). When all pairs had been compared with each other, the results were displayed to the respondent by a computer in order to verify the preference order and the relations between the criteria. If the results did not correspond to the respondent’s views, the weights were modified accordingly.

The results of the weightings were processed using the Hipre software in order to elucidate the method and to present the

preliminary results in the seminar. The weights scaled by Hipre were also used as inputs in our DST.

A systematic procedure for compiling the valuation results is needed if multiple experts are involved in defining the weights. There are several methods to aggregate the individual weights. These include using the weight assigned by the largest number of respondents (majority criterion), extreme values, calculated mean or ratios of weights (Belton and Pictet, 1997; Rogers and Bruen, 1998). Arithmetic mean is the most common method of combining a set of weights and several studies indicated that it is a feasible approach (Meyer and Booker, 1990). We chose to use both aggregated weights corresponding to the arithmetic mean values and individual weights for calculating the preference scores. In the latter case, the RM alternative that received the highest preference score from the largest number of respondents was identified as the preferred one.

2.5. Sensitivity analysis

It is a well-known feature of hierarchical multi-attribute models that the weights of the factors at the highest level (i.e. criteria) in the hierarchy have the greatest impact on the final preference score, while the effect of the variation at the lower levels (i.e. levels including attributes or sub-attributes) generally results in a much

diminutive influence (e.g. Hämäläinen and Lauri, 1992; Butler et al., 1997). Therefore, to study the effect of the variability of weights on the total preference scores, we carried out a 'one-dimensional' sensitivity analysis by separately varying the single weight of each criterion while the original ratios between the weights of other criteria were kept constant. This analysis made it possible to find turnover points of weights where the ranking of the remediation alternatives changes in our model sites.

3. Results

3.1. Weights set by the stakeholders

The weights set by different people varied considerably resulting in slightly different preference scores of the RM alternatives (Table 5). This was expected, since the weights reflect each person's individual values and attitudes, personal and professional history, education, cultural background, knowledge level, the stakeholder group he/she represents etc. The differences may also result from some misunderstandings in the weighting task (see Section 4.3).

The rough comparison between the weights based on ratio estimation versus pair-wise weighting showed that the different techniques incur slightly different weights and consequently, different preference scores (see below Section 3.2). The different results can also manifest some difficulties in the valuation. It should be noted that our result is based on very limited material since only six persons carried out both weightings. Hence, it is not possible to draw any definite conclusions on the validity of the weighting methods. Moreover, only the criteria were valued using the pair-wise weighting. It is possible that the weighting of attributes and sub-attributes too, would have resulted in wider variation between the final preference scores.

3.2. Preferred risk management alternatives

The results based on the use of the aggregate weights, i.e. mean values calculated from the respondents' individual weights, show the preferred RM alternatives for the model sites (Fig. 2).

In the shooting range study, the RM alternatives referring to soil washing (Alt. IV) and land use restrictions with groundwater treatment at waterworks (Alt. V and VI) gained almost equal total preference scores. 'Costs' and 'Risk reduction' were clearly the most important decision criteria. Under the criterion 'Environmental effects', the attributes 'waste generation' and 'soil loss' were the most predominant.

In the case of the gasoline station, the Monitored Natural Attenuation (MNA) method combined with soil vapor extraction (SVE) turned out to be the most preferred RM alternative (Alt. IV). Besides 'Costs' and 'Risk reduction', the criterion 'Environmental effects' came across as a significant factor contributing to the final preference score. Here, the major attributes affecting the value of the latter criterion were 'space use' (area unusable due to contamination or ongoing remediation activities), 'waste generation' and 'soil loss'.

When we examined each respondent's individual preference scores, in the case of the shooting range only two alternatives came up as the preferred RM option (Table 6). These two RM alternatives were also among the three alternatives that received the highest preference scores when we used aggregate weights in the calculations. In the case of the gasoline station five alternatives emerged including the 'no remediation' alternative (Alt 0). In other words, the individual preference scores differed significantly from the preference score calculated on the basis of the mean weights. The grounds for these differences are discussed in Section 4.3. Despite these differences, using the individual weights produced exactly the same preferred remediation alternative as using the aggregated weights. Furthermore, the two different weighting techniques gave almost equivalent results in the case of five respondents out of six (Fig. 3).

It is noteworthy that particularly in the case of the shooting range the expected risk reduction in health risks and ecological risks was very high in all RM alternatives, Alternative 0 (no remediation) being the only exception. Therefore, there were only slight differences between the different alternatives in the final value scores of the criterion 'Risk reduction'.

The results of the sensitivity analysis for both model sites show that the ranking of the RM alternatives is quite sensitive to changes in the criterion weights (Fig. 4). On the other hand, the best RM alternatives seem to be quite stable towards small changes in the weights around the mean values.

Table 5
Variation of the weights given by different respondents: statistics of the scaled weights (unitless) of the criteria (p_c), attributes ($w_{c,i}$) and sub-attributes ($w_{c,i,i}$). See Fig. 1 for the hierarchy between the criteria, attributes and sub-attributes. STD = standard deviation, n = number of respondents.

Criterion	Shooting range ($n=19$)				Gasoline station ($n=28$)			
	Mean	STD	Min	Max	Mean	STD	Min	Max
p_1 : Risk reduction	0.36	0.15	0.05	0.76	0.29	0.16	0.03	0.71
p_2 : Environmental effects	0.23	0.14	0.04	0.53	0.26	0.10	0.06	0.48
p_3 : Costs	0.30	0.12	0.05	0.48	0.32	0.17	0.07	0.63
p_4 : Other factors	0.11	0.09	0.02	0.38	0.13	0.08	0.03	0.30
<i>Attribute</i>								
$w_{1,1}$: Health risks	0.35	0.16	0.09	0.63	0.35	0.23	0.04	0.94
$w_{1,2}$: Ecological risks, terrestrial	0.21	0.15	0.03	0.05	0.31	0.15	0.01	0.56
$w_{1,3}$: Groundwater quality	0.44	0.13	0.22	0.71	0.34	0.16	0.05	0.71
$w_{2,1}$: Emissions to air	0.13	0.10	0.00	0.33	0.14	0.09	0.00	0.35
$w_{2,2}$: Energy consumption	0.15	0.10	0.01	0.38	0.13	0.07	0.01	0.29
$w_{2,3}$: Soil loss	0.25	0.09	0.08	0.42	0.16	0.09	0.03	0.40
$w_{2,4}$: Groundwater loss	–	–	–	–	0.08	0.06	0.00	0.26
$w_{2,5}$: Space use	0.13	0.07	0.04	0.36	0.18	0.14	0.05	0.70
$w_{2,6}$: Waste generation	0.34	0.14	0.14	0.65	0.31	0.19	0.08	0.86
$w_{4,1}$: Ecological impact	0.68	0.17	0.33	0.91	0.38	0.21	0.09	0.91
$w_{4,2}$: Image aspects	0.32	0.17	0.09	0.67	0.62	0.21	0.09	0.91
<i>Sub-attribute</i>								
$w_{2,6,1}$: non-hazardous waste	0.21	0.13	0.04	0.56	–	–	–	–
$w_{2,6,2}$: heavily contaminated soil	0.31	0.08	0.14	0.43	0.52	0.23	0.17	0.91
$w_{2,6,3}$: hazardous waste	0.34	0.16	0.06	0.71	0.48	0.23	0.09	0.83
$w_{2,6,4}$: wastewater and sludge	0.15	0.08	0.03	0.38	–	–	–	–

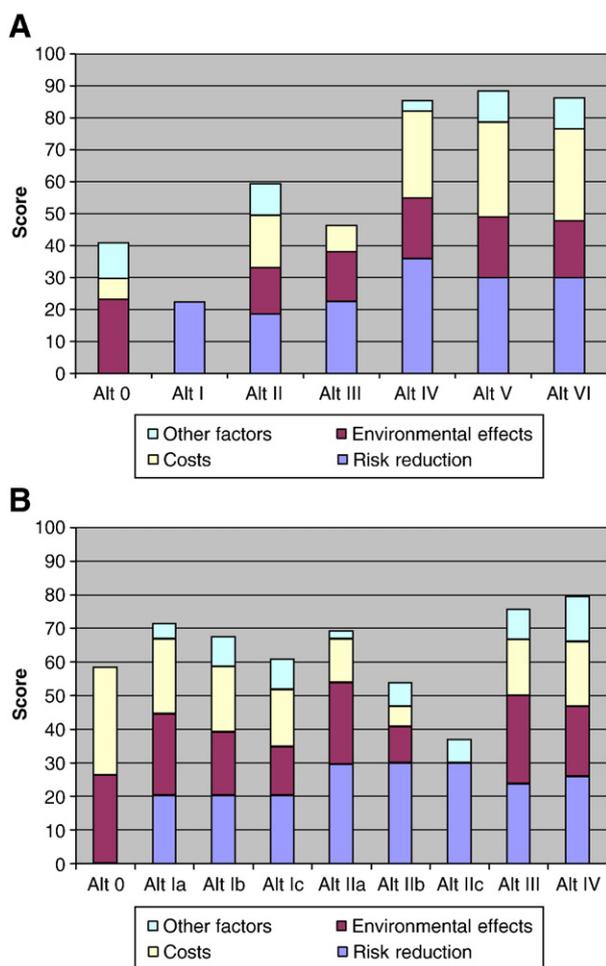


Fig. 2. Preferences for the alternative risk management (RM) methods (the RM alternatives are described in detail in Table 2) of the model sites: shooting range (A) and gasoline station (B), and the contribution of each criterion to the total preference score.

In the case of the shooting range, ‘Costs’ seems to be a totally indifferent factor. Whereas altering the weight of the criterion ‘Risk reduction’ only affects the mutual order of the three most preferred RM alternatives. Furthermore, either the criterion ‘Environmental effects’ or ‘Other factors’ should gain a weight higher than 0.8 in order to supersede the two preferred alternatives. In the case of the gasoline station, the best RM alternative (Alt IV) is changed (to Alt 0 corresponding the

Table 6
The influence of the variation of individual weights on the preference of the risk management (RM) alternatives. Share = proportion (%) of the respondents who prioritized the RM alternative as the most preferred based on their weights.

Shooting range		Gasoline station	
RM alternative	Share, %	RM alternative	Share, %
Alt 0	0	Alt 0	19
Alt I	0	Alt Ia	12
Alt II	0	Alt Ib	0
Alt III	0	Alt Ic	0
Alt IV	35	Alt IIa	15
Alt V	65	Alt IIb	0
Alt VI	0	Alt IIc	0
		Alt III	15
		Alt IV	38

option “no remediation”) if the weight of the criterion ‘Risk reduction’ decreases from 0.26 to 0.2 while the original ratios between the other weights remain constant. Increasing the weight of the criterion ‘Costs’ to around 0.6 has the same effect, whereas the weight of the criterion ‘Environmental effects’ has to be above 0.4 in order to alter the preferred RM alternative. Moreover, even if the weight of the criterion ‘Other factors’ is varied, Alt IV remains the best RM option.

4. Discussion

4.1. Selection of aggregation methods

We decided to use MAVT as the aggregation method in our study. The main justification for this selection was consistency since the calculation of the index depicting the environmental effects was based on MAVT. However, there are several other aggregation methods that could be used as a starting point. According to Guitouni and Martel (1998) compensation degree is one of the key aspects in the selection of the method. Any MCDA method can be classed as being compensatory, non-compensatory or partially compensatory. MAVT can be considered to be partially compensatory meaning that some compensation is accepted between the different decision criteria but a minimum level of performance is required from each of them. For example, in our case this could mean that low costs can compensate low risk reduction in any RM alternative. In reality, the decision-makers might be unwilling to accept such tradeoffs. In these cases non-compensatory MCDA methods, such as ELECTRE (based on the identification of dominance relations) would be most suitable. It could therefore be useful to study the applicability of other aggregation methods to our study problem.

The use of the arithmetic mean in aggregating the individual weights has been criticized in some studies. For example, Koffler et al. (2008) state that one of the main shortcomings of this method is that it is blind to the individual’s preferences towards other criteria. Therefore, these researchers recommend that the individual weights are preserved and carefully regarded in the MCDA procedure. In our study, this aspect was taken into account by using both individual weights and aggregated weights.

4.2. Components of our DST and comparison with other DSTs

The decision criteria and outcomes of our DST slightly differ from those of the Dutch REC system and the other existing DSTs. Compared with REC our DST includes an additional criterion ‘Other factors’ that comprises social aspects, among other things. However, only image aspects were considered in our case studies since other social impacts were considered insignificant. Many existing DSTs ignore the social aspects and in those DSTs where they are involved, the focus is normally only in socio-economic issues (e.g. Carlon et al., 2007; Cox and Crout, 2003; SMARTe, 2009). Hence, in many DSTs social aspects are dealt with using economic indicators i.e. they are monetized. By contrast, in our DST monetization is not used to quantify social factors.

Since our DST was originally based on the REC system it is basically very similar to it. However, there are also some principled differences between these two DSTs. Firstly, a value tree was the starting point for both DSTs. Moreover, equivalent to REC, our DST is built in Excel and it includes separate modules that represent the decision criteria. Like the REC_{Urban} tool our DST does not include equations for the calculation of health risk estimates. In practice, the choice of risk assessment methods depends on the study problem, available input data and the expected accuracy of the results. By necessitating the use of a separate tool, we wanted to stress the importance of expertise in the selection of the method and interpretation of the results. Moreover, although REC includes equations for evaluating runoff, leaching of some contaminants into groundwater and plant uptake,

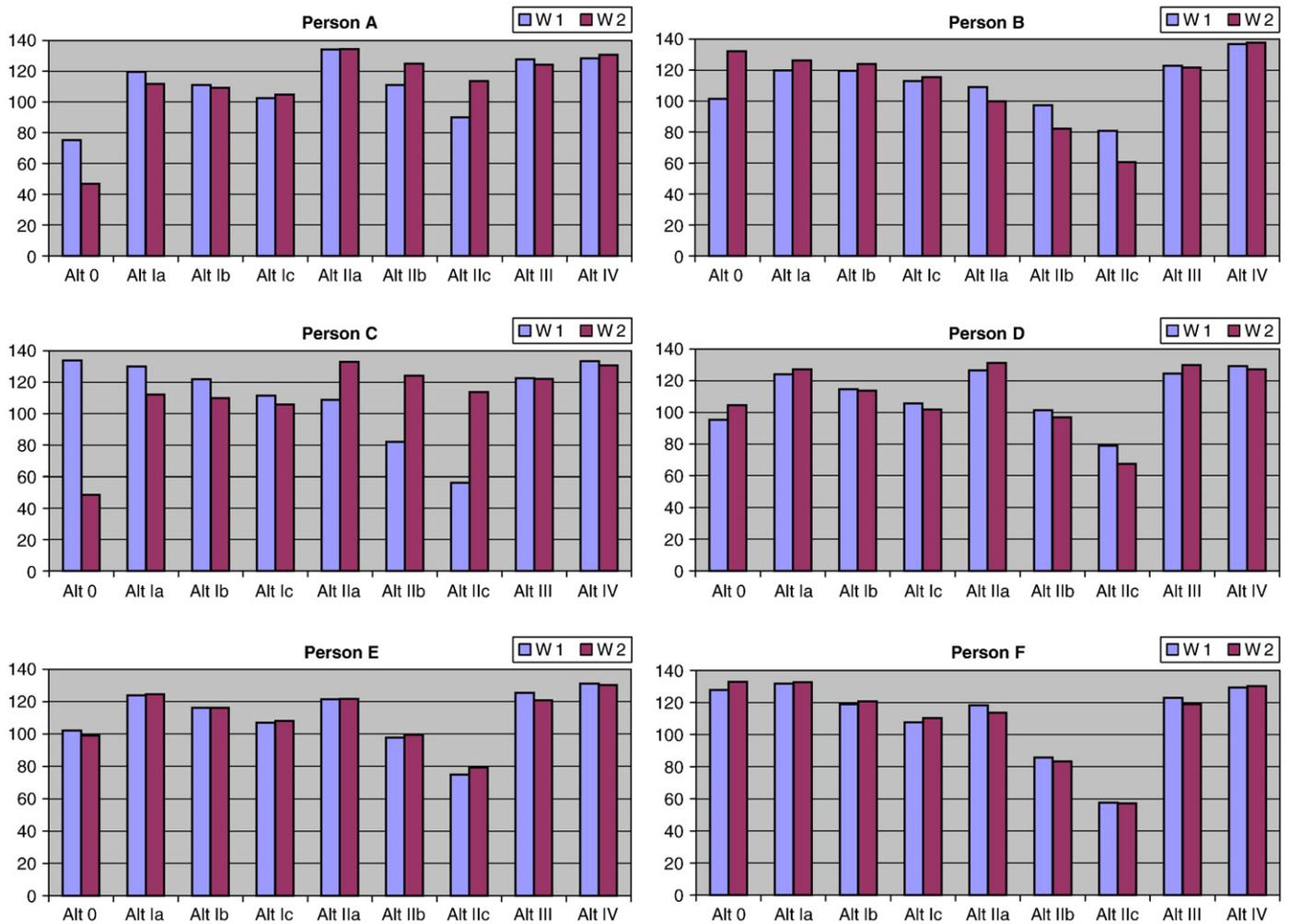


Fig. 3. Effect of the weighting method: scaled preference scores of the alternative risk management (RM) methods of the gasoline station based on the weights defined by six persons (A...F). To make the results commensurable a scaling was conducted by multiplying each preference score of a particular RM alternative by factor 1000 and dividing it by the sum of the preference scores of all RM alternatives. W 1 = weighting based on ratio estimation, W 2 = weighting based on pair-wise comparison.

these we not included in our DST because we had no information on their applicability to Finnish conditions. We also wanted to stress the importance of in situ or laboratory-scale studies and the use of more detailed transport models.

Unlike in REC, the criterion 'Environmental effects' in our DST only includes negative environmental factors, and hence, it does not embrace the factors 'soil quality' and 'groundwater quality'. Instead, a separate attribute 'groundwater quality' was added under the criterion 'Risk reduction'. Furthermore, the values for the attribute 'Emissions to air' in our DST are calculated using a different life cycle based impact assessment method than in REC. In addition, in our DST the ranking of environmental effects is based on a case-specific approach instead of using a generic reference like in REC (see below). Moreover, the outcomes of our DST include aggregated preference scores, which can be used to quickly compare different RM alternatives.

It is noteworthy that the weights in our DST are always associated with the particular data involved in the RM alternatives, i.e. our solution is based on a case-by-case evaluation that is a typical situation in the application of DSTs (e.g. von Winterfeldt and Edwards, 1986). In contrast, in REC a Dutch average remediation case is used as a reference in the determination of environmental effects (Beinat and van Drunen, 1997; van Drunen et al., 2005). This leads to a solution in which the weighting factors reflect the values of the reference. However, this solution requires that the DST

analyst³ is capable of measuring the attribute values of a new case study in a way comparable with the reference. According to our experience, this task is difficult to carry out due to the lack of data and scientific knowledge of land contamination, and the variability of sites.

4.3. Notes and feedback from the weighting process

In the context of DSTs, it is assumed that the criteria and (sub-) attribute weights are directly derived from a group of people (panel) by elicitation. Elicitation is a process of gathering judgments concerning the decision problem through specific methods of verbal or written communication (Meyer and Booker, 1990). It is generally known that individual weights determined on the basis of individual valuation differ considerably, partly due to different opinions, and partly due to biases originating from the behavior of the experts, and the procedures and techniques used in the elicitation. According to a summary of Seppälä (2003), the factors causing different results in the weighting process are: the composition of the panel, the format of questions, available

³ The term DST analyst refers to the expert who determines or is heavily involved in the determination of the values for the attributes and sub-attributes and acts as a moderator in the weighting process. The role of the analyst assumes adequate knowledge of the methods applied in decision analysis and risk assessment.

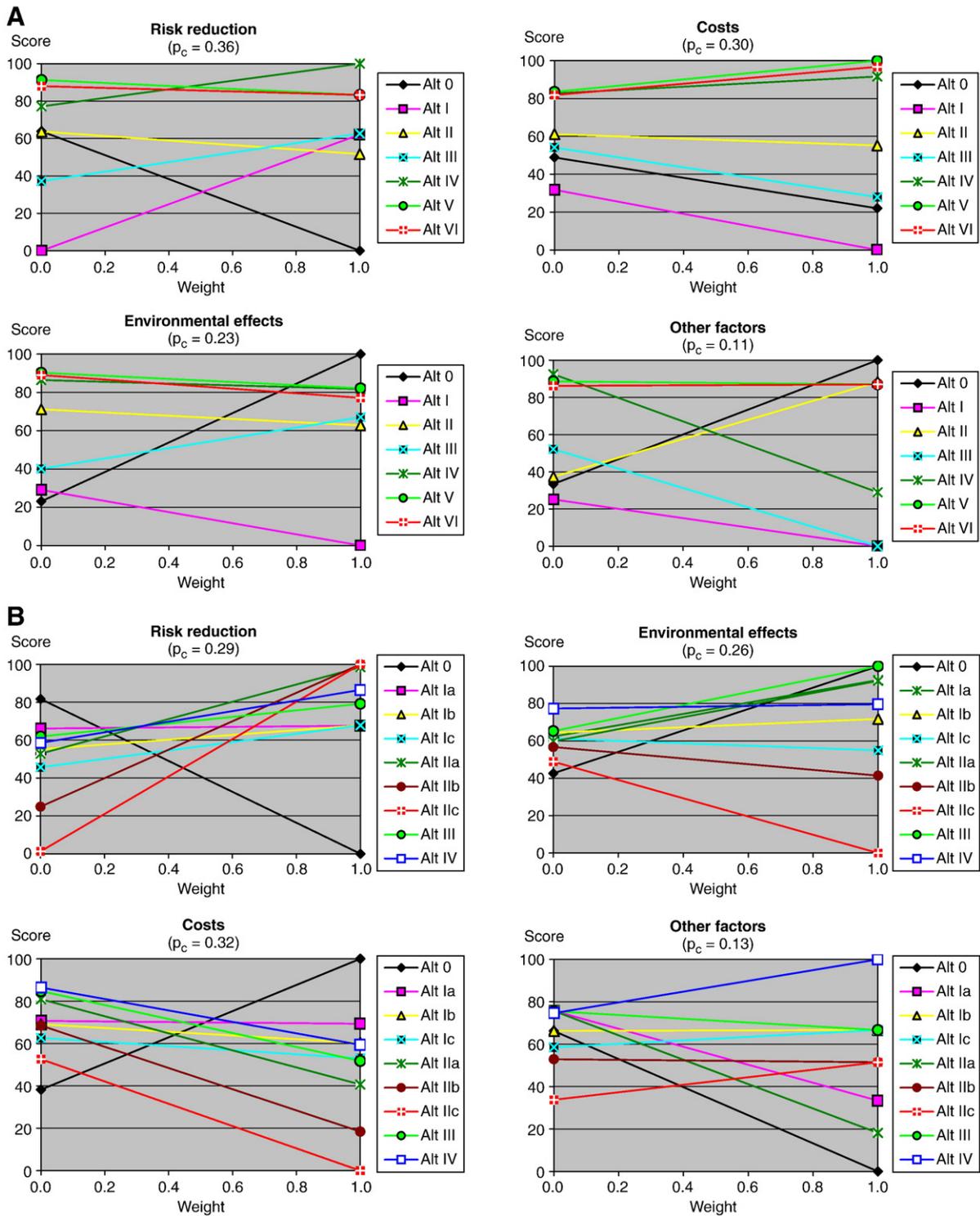


Fig. 4. Sensitivities of the preference scores of the risk management (RM) alternatives to changes in the criterion weight in the case of the shooting range (A) and gasoline station (B). The chart illustrates the changes in the ranking of the RM alternatives along the variation of the weights of individual criteria. The value in parentheses corresponds to the original aggregated weight (i.e. the arithmetic mean calculated from individual weights) of the particular criterion and forms the starting point of sensitivity analysis. The upmost line represents the preferred RM alternative determined by the values and the particular set of weights of the criteria.

information, criteria applied, weight elicitation techniques and the calculation techniques of weights. Some problems related to these aspects also emerged during the demonstration of our DST.

First of all, the valuation of some factors was regarded as somewhat problematic due to the difficulties of comparing them, e.g. the attribute 'waste generation' against 'emissions to air' or 'space use'. Therefore, it may be necessary to develop these attributes more comparable with

each other. One way of making all criteria and attributes comparable with each other is to monetize them. Economic values already exist for health risks, risks to biota and environmental load. Other factors, e.g. other ecological values and social factors, could also be monetized using different techniques such as Contingent Valuation, Hedonic Price Method or Avoided Cost Approach (e.g. Pethig, 1994). However, while monetization is often preferred by economists and the method might

be handy particularly when studying the cost-efficiency of remediation alternatives, it is not necessarily feasible in the case of decision-making involving multiple and originally incompatible criteria. In fact, Bardos et al. (2002) state that the possibility of not being forced to monetize all factors involved can be considered the merit of MCDA methods in the CLM context. The infeasibility of monetization is also manifested in the fact that some aspects, which could be important to an individual stakeholder, will be lost. Some people might also find it unethical or incomprehensible to measure human life, well-being or environmental values in terms of money. Nevertheless, in the future it would be worthwhile to test the process of monetization for the determination of the weights in order to avoid biases in valuation.

Some respondents considered it difficult to comprehend the magnitude of some criteria, attribute, or sub-attribute values in the context of the model sites (e.g. the sub-attributes 'emissions to air' and 'energy consumption' characterized by the unit 'inhabitant equivalent'). Some individuals' weights also deviated considerably from the other respondents' corresponding weights giving grounds to doubts that misunderstanding had occurred. However, since it was impossible to indisputably justify this interpretation, we did not eliminate such outliers from the calculations. The deviations in the preference order may also have partly arisen from the different scaling of the preference values (see below).

The ratio estimation technique is a simple valuation method. On the other hand, it is somewhat unclear how people understand the 'ratios'. Therefore, individual scaling is often a problem in weighting tasks when individual answers are combined to produce an aggregate group response. This fact also emerged in our study: while some people used, for example, a scale from 10 to 50, others used a scale from 10 to 1000 to indicate the weights between different criteria/attributes/sub-attributes. The problems of the predominance of the wide scale in the final aggregate weight and its manifestation as the considerable variation of the weights have been identified in many studies (e.g. Seppälä, 1999).

The participants' specific expertise was reflected in the weighting process. For example, persons representing the land owners tended to assess their individual preferences of some attributes and sub-attributes on the basis of cost effects. To give an example of this, when weight was given to the attribute 'waste generation', which is under the criterion 'Environmental effects', they tried to value the attribute on the basis of the costs of waste disposal or treatment instead of environmental aspects. Hence, it is obvious that the basis of valuation has to be stressed throughout the weighting process. It also proved necessary to emphasize that when valuating the criteria, the attributes and the sub-attributes related to the criterion have to be kept in mind. For example, when the criterion 'Environmental effects' is valued at the ranges of the sub-attributes, i.e. 'emissions to air', 'energy consumption', 'soil loss', 'groundwater loss', 'space use' and 'waste generation', have to be considered. In complicated case studies with a significant amount of data, it is difficult to keep all the data in mind when setting the weights.

It is also noteworthy that the temporal scope of the consequences associated with the RM actions is often an important decision criterion. In our DST (and also in REC), the time aspect is not considered separately but is included in the calculation of values for the attributes under the criteria 'Risk reduction' and 'Other factors' and the sub-attribute 'space use' (under the criterion 'Environmental effects'). In the calculation of costs, the time span is considered by discounting. Since the time aspect is more or less 'hidden' in the calculations, in the case of factual decision-making it is often important to also study the different RM options separately from the viewpoint of the expected time needed to reach the final target risk level or the point when the costs or other negative or positive impacts occur. In fact, in our model sites the preference of the RM alternatives with a long time span such as MNA can be partly explained by the fact that the time aspect was not explicitly included in the criteria and attribute values.

While there are indisputable benefits of using multiple criteria DSTs, some projects abroad have also shown limitations in such methods. These problems appeared when negotiating parties had different value systems (e.g. Page et al., 1998). Such situation could occur in the case of CLM where several stakeholders representing different fields and personal and professional background are involved. Consequently, it can be difficult to agree on weighting of the factors involved and some stakeholders might be hesitant to engage the valuation exercise. However, such problems were not identified in our study.

4.4. Uncertainty involved in the attribute values and in the value tree

The results from our demonstration using two model sites are hampered by some uncertainties mainly owing to the lack of data and the characteristics of the DST (Table 7).

First of all, we assumed that the value functions of all attributes and sub-attributes were linear. In practice this is not necessarily the case. We chose the linear value functions because they allow a simple solution for the description of the preferences of attribute values. However, in the future it is worthwhile to test the use of non-linear value functions in our DST model.

Other uncertainty factors include the variability and uncertainty of the data particularly related to the costs and risk estimates, which all have a major effect on the final preference scores. In practice, these uncertainties mainly arise from the inability to accurately define the scale of contamination or in some cases, from the uncertainties associated with remediation methods. Improper risk assessment methods can also lead to unrealistic risk estimates. However, the uncertainty coming from these should be minor since only the relative risk reduction is considered in the calculations. The lack of accurate data is a problem particularly in the case of novel remediation techniques such as MNA (gasoline station), reactive barrier (shooting range), Metclean (shooting range) and membrane filtration (shooting range). Since the main focus of this study was to test and demonstrate the usability of our DST, the uncertainties in the values of the attributes and sub-attributes were not assessed quantitatively.

In our study, we did not consider different structures of the value tree. In practice, different structuring of the value tree can result in different weighting results and consequently, varying preference scores. This can appear as a higher weight if an attribute/sub-attribute is located higher in a value tree or as splitting bias (e.g. Pöyhönen and Hämäläinen, 1998). Splitting bias refers to a phenomenon in which the overall weight of an attribute is the higher the more there are sub-attributes in a branch of that attribute in the value tree. Hämäläinen and Alaja (2006) proved that splitting bias was systematic but not a problem among engineering students. While in the case of laymen it was a true issue. The authors also point out that hierarchical weighting (followed also in our study) instead of non-hierarchical is a potential way to eliminate splitting bias. In the case of our DST, the presence of splitting bias particularly in the weighting of the attribute 'waste generation' may be worth of studying. Otherwise, we consider the reasonable options for the value tree to be very limited. Hence, we expect the effect of splitting bias to be quite minor.

4.5. Applicability and usability of the DST

The ranking of RM alternatives in our DST is based on the traditional decision analysis allowing its versatile and case-specific use. One of the main outcomes of our DST is an aggregate value score (preference score) that enables a simple and fast overall comparison of RM alternatives. In addition, the major assets of our DST include its full transparency, flexibility, convertibility and the possibility to connect it with other Excel-based calculation tools such as Crystal Ball or @Risk (statistical software tools) and CalTox (a tool for calculating human health risks, freely available at <http://eetd.lbl.gov/ied/ERA/caltox/>). Transparency means that all calculation methods and default

Table 7

Summary of the main uncertainties involved in the preference scores. TPH = total petroleum hydrocarbon, BTEX = benzene, toluene, ethylbenzene, xylenes, TVOC = total volatile organic compounds; RM = risk management. + = increases preference score, – = decreases preference score, ? = effect unknown.

MCDA component	Effect on the criterion- specific preference scores	Uncertainty aspects
Form of the value functions Cost estimates	?, varies depending on the criterion/(sub-)attribute +/-, depends on the risk management method	Value functions were assumed to be linear. In the excavation option, the volume of soil is critical; reliability of the estimates is a particularly relevant issue in the case of new remediation methods (see below).
Data (low reliability) on the new remediation methods	–	The evaluation of the attainable risk reduction and costs was based on – A single Finnish experimental project – A single data source, method has not been used for Pb removal in Finnish waterworks .
– MNA, reactive barrier – Metclean, membrane filtration	gasoline station: Alt III, IV shooting range: Alt V, VI	The site-specific data only include data on TPH, BTEX and TVOC. Since toxicity reference values only exist for BTEX, the risk estimates considerably underestimate the actual risk levels.
Health risk assessment, gasoline station	?	In the case of MNA and composting, the residual risks are probably underestimated due to faster degradation of BTEX compared to the heavier TPH fraction.
	– (Alt Ic, IIc, III, IV)	
Attribute values under the criterion 'Other factors'	?	We used our own judgment based on the characteristics of the sites
All weights		Our preliminary study showed minor effects on the individual weights.
– weighting technique	+/-	
– weighting process	?	The accuracy of weights is diminished by several factors (see Section 4.3 and 4.4)
Weights for attributes under the criterion 'Environmental effects'	?	Problems were encountered in the direct weighting of some incompatible attributes (see Section 4.3)

input data are documented, whereas full flexibility and convertibility means that criteria, attributes and sub-attributes can easily be added into and eliminated from our DST. This allows using heterogeneous data (similarly to the REC system), including qualitative information, with varying levels of elaborateness. Unfortunately, convertibility also increases the risk of misusing the DST since the principles of the calculation methods need to be understood. Therefore, the use of expertise is necessary if modifications are needed.

Our model sites used in testing and presenting the DST to the invited RM experts were deliberately created to be as simple as possible but to still represent realistic cases. Therefore, we did not consider combinations of different remediation techniques within a single RM alternative. In practice, a single RM option often includes several remediation methods. In such cases, using the DST for determining the most eco-efficient and/or preferred RM actions may require dividing the site into sub-sites as per the RM options. In fact, in such cases using the DST can bring the highest value to decision-making since it can be difficult to identify the best RM option without using a systematic, mathematical approach.

Another simplification in our study compared to actual contaminated sites in Finland was the assumption that contamination was only caused by a single contaminant or several similar contaminants that can be treated as 'one compound' (e.g. petroleum hydrocarbons). However, even in the case of multiple chemicals the key contaminants can be identified using for example scoring systems (USEPA, United States Environmental Protection Agency, 1989). Then minimizing the risks arising from these becomes the main goal of the RM actions and therefore, the number of available RM options will be more limited and the problem will be simplified from the viewpoint of using the DST.

It is noteworthy that producing all the data needed for using our DST requires expertise. However, such expertise is needed when selecting RM methods even if the DST were not used. First of all, since our DST allows the use of any risk assessment methods, no additional data are needed for the 'Risk reduction' module of the DST. According to the Finnish legislation it is compulsory to conduct a risk assessment when remediation need is determined (Ministry of the Environment,

2007), the methods, however, can be selected case-by-case. Secondly, since the criterion 'Costs' is obviously the key factor in every CLM decision, the cost data should be readily available. The current environmental legislation also assumes the consideration of economic aspects when deciding on the RM actions (Ministry of the Environment, 2000 and Ministry of the Environment, 2008). Data on the environmental effects (e.g. emissions and wastes generated) can be the most difficult to attain, however, the provider of a particular technology is liable for providing these. Our DST also includes such data on several remediation methods. Lastly, no specific data are needed for the evaluation of social and other adverse effects assessed within the criterion 'Other factors'. Determining the values for these attributes requires some understanding of the potential effects of different RM options but can normally be carried out e.g. by a group comprising different stakeholders (such as CLM experts and authorities). Setting accurate values for some of the attributes assumes using methods applied in social sciences. However, both the results from our seminar and the feedback from the recent project which used our DST to assess the preference of various RM alternatives at two actual contaminated sites (Lunden, 2008), speak for the usability of the simple scaling method adopted in our DST.

It needs to be emphasized that only those criteria and (sub-) attributes that are relevant and at least to some extent conditional in a particular RM case, and the true RM alternatives for which no clear preference can be found should be included in the analysis using the DST. This allows optimizing the resources and collecting of unnecessary data, i.e. data that is not profiting decision-making, is avoided. It is also noteworthy that carrying out the weighting procedure requires expertise and assumes proper planning and advance arrangements. Therefore, feasibility of the weighting task should be assessed case-by-case.

5. Conclusions and future prospects

The decision support tool (DST) we developed for prioritizing risk management alternatives for contaminated sites is based on the decision analysis framework in which the elements of the preference model were established based on the multi-attribute value theory

(MAVT). The final tool allows a systematic comparison of different RM alternatives and determination of their eco-efficiency or cost-efficiency. The DST is particularly useful if none of the optional RM actions can be clearly prioritized. Furthermore, the framework used in the DST makes it possible to identify and consider the preferences and subjective views of different stakeholders (e.g. risk managers and authorities) in decision-making. Moreover, the DST facilitates communication and information exchange between the stakeholders, and provides means for public participation. This way conflicts that could delay RM actions may be avoided.

Our DST is more case-specific compared with the Dutch REC system, from which its basic elements were derived. In our DST, the preference scores for alternative RM options are calculated using the weights determined for the factors i.e. decision criteria and their attributes and sub-attributes involved. The weights should be set site-specifically taking into account the numeric values of the criteria, attributes and sub-attributes; type, magnitude, and scale of contamination; land use; and environmental conditions. Due to the site-specificity, the results from our demonstration using two model sites are not straightforwardly applicable to other situations. However, the weights that were defined could be adapted in the case of equivalent sites (e.g. gasoline stations).

The demonstration of our DST by two model sites showed that attention should be paid to proper and detailed problem formulation including exact processes for eliciting weights in order to avoid misinterpretations and misunderstandings. Using different weighting techniques (i.e. ratio estimation and pair-wise weighting) and alternative ways to treat individual respondents' weights in calculating preference scores can provide additional information on the consistency of the ranking of RM alternatives.

While our concise review of some existing DSTs was only focused on the generic structural and functional properties, it might be worthwhile to conduct a more detailed comparison study and to include additional DSTs in it in order to find ways to develop our DST more comprehensive. As suggested by Agostini et al. (2009), in such study the advantages and disadvantages of different tools could be revealed by using them for solving the same decision problem. However, in the first instance we intend to complement our DST with a generic risk assessment module, which could be used as a screening level tool to determine human health risks. Furthermore, the possibility to add contaminant transport models (soil erosion and leaching) representative to Finnish conditions needs to be studied. There is also an ongoing project in SYKE where a simple, generic tool is being developed for the screening level selection of the best available remediation technologies. It would be useful to link this tool with our DST. We also plan to include statistical methods in order to consider the uncertainty and variability of the attribute and sub-attribute values since in practice these are critical factors contributing to the final preference scores.

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