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INDICES OF SOIL QUALITY: A MULTICRITERIA VALUE FUNCTIONS APPROACH

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ABSTRACT

This article addresses the estimation of soil quality for evaluating clean-up alternatives for polluted soils. In many practical applications the goal of a high soil quality after clean-up cannot be reached due to technical or budget constraints. The selection of the cleaning-up technique requires trade-off analyses between soil quality and other features. Soil quality measures are based on dose-effect functions, which translate contaminant concentrations into effects. Since dose-effect functions are not available for most common pollutants, substitute evaluations based on expert judgements are used. The article addresses the substitution of soil quality indices with multicriteria value functions assessed through expert judgement. It describes a method and a software package designed for this purpose and shows the use of the system in real applications.

INTRODUCTION

Soil contamination has become a great concern in The Netherlands. In 1983 the Dutch Government has released the soil clean-up guideline containing administrative guidelines for the implementation of the Dutch Soil Clean-up Interim Act [1]. The guideline aims at setting criteria to evaluate the urgency of sanitation and remediation techniques for contaminated sites. Revisions to the guideline have been recently published [2] and amendments to the Act are due to be in force in the near future [3]. The guideline fixes basic rules for priority setting and

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objectives for sanitation operations, but does not include a detailed method to select the best clean-up technique on a given site.

Determination of appropriate clean-up strategies is difficult due to the complexity of the factors involved, such as kind and type of contaminant, pollutant effects, soil characteristics, and land use. This highlights the need for a systematic approach that supports cost-effective choices and that increases transparency and uniformity in decision making.

DECISION MAKING STRATEGIES FOR SOIL SANITATION

A cornerstone of the guideline is that a site after sanitation should be suitable for every possible future activity. The soil quality should be sufficient to pose no harm to humans and ecosystems and to avoid contaminating other environments. This principle is called "multifunctionality of the soil" and should be the objective of every remediation process. The end goal of the clean-up process, the soil quality level, is evaluated against standard concentrations or target values, such as the Dutch list of A, B and C standards [1, 2]. The A reference value indicates clean soil while the C value is the trigger concentration for clean-up operations. The B value represents an intermediate situation, where additional information on the nature, place, and concentration of the contaminants is necessary before deciding on cleaning-up operations.

Objectives of the Clean-up Process

Clean-up processes aim at restoring soil multifunctionality reducing contaminant concentrations below the A values. However, the achievement of this objective can be hampered by budget and technical constraints.

It has been estimated that the number of Dutch sites which need remedial actions within twenty-five years exceeds 100,000 [4, 5]. Considering that the cost of cleaning-up a ton of contaminated ground can go up to 650 U.S.$ [6], the total clean-up costs are estimated to be at least 25,000 million U.S.$ [5]. In recent years, annual expenditures for remedial operations can be estimated around 300 million U.S.$. Although clean-up technologies are likely to improve in the near future and determine significant cost reductions, clean-up operations imply an enormous expenditure at the national level and budget constraints are likely to play an important role at the local level.

In addition, research studies based on the 1990-1992 period "conclude that treatment results have certainly improved in the last while; however, in spite of high efficiency, only about 50% of the cleaned soil meets the Dutch standards (A-values)" [3].

Under these conditions it is often necessary to accept a compromise between ecological and economical concerns and implement remediation techniques which do not fully meet the multifunctional goal [7]. Once it has been decided to clean up a site, the selection of the most suitable technique requires a detailed analysis of the trade-offs between the objectives of limiting the costs and achieving high environmental quality.

This is the typical domain of Multiple Criteria Decision Analysis (MCDA), which provides tools to structure the decision process and helps the decision maker find the most satisfactory balancing between conflicting objectives [8-10]. A specific application to the soil clean-up process is the SOILS decision support system, which delineates a systematic approach to the evaluation of cleaning-up alternatives and supports the selection of the most suitable technique [11].

Multiple Criteria Analysis for Evaluating Cleaning-up Alternatives

The evaluation framework used in SOILS requires the alternatives to be compared on the basis of multiple criteria, such as clean-up costs, residual concentrations of contaminants, time of operations, nuisance, etc. The estimated performances of clean-up alternatives are organized in an "Effects Table," as shown in Table 1.

The selection of the best technique is based on a multicriteria value function model. The model assigns a value index to each alternative synthesizing global performances into a numerical score. The highest score corresponds to the best possible combination of performances, while the lowest score represents the worst possible combination of performances.

THE VALUE FUNCTION MODEL

The value index attached to each alternative is the combination of elementary indices calculated for each criterion. The weighted additive is the simplest value function model.

Table 1. Example of Effects Table for Three Alternatives and Three Criteria

	Alternatives		
	A_1	A_2	A_3
Cost (1000 Dutch Guilders)	10,000	30,000	22,000
Residual pollutant concentration (mg/Kgds: mg pollutant/Kg dry soil weight)	150	100	200
Sanitation time (days)	65	80	25

To specify the weighted additive model it is necessary to go through four major steps [12]. Without loss of generality it is assumed that preferences over criterion scores are monotonic, either increasing or decreasing, i.e., the higher (lower) the score, the higher (lower) the preference.

The first step requires the definition of the range of scores for each criterion. If X_i and R_i, i = 1,...,n, indicate the evaluation criteria and the ranges, the performance score x_i on X_i is such that $x_i \varepsilon R_i$. For a soil pollutant, for instance, the range could be delimited by a very low concentration representing a clean soil and a very high contaminant level representing unacceptable pollution. These limit situations, marked with x_i^* and x_{i*}, i = 1,...,n, are assigned the 100 and 0 limit values, where 100 is the best and 0 the worst performance within the range.

The second step is the assessment of the value functions for each criterion. They associate with each score a value representing the preference in comparison to the limit situations. In symbols, given $v_i(.)\!:\!R_i \rightarrow [0,100]$ the value function attached to X_i, it is such that x_i' is preferred to x_i'' if and only if $v_i(x_i') \geq v_i(x_i'')$, for any $x_i', x_i'' \varepsilon R_i$, i = 1,...,n. Figures 1 and 2 show examples of value functions for lead soil pollution and sanitation time respectively.

The value function in Figure 1 is associated with the effects of the pollutant, measured on a relative scale from 0 to 100. The best value corresponds to the absence of contamination, whilst the worst corresponds to a very high concentration typical of a severely polluted soil. Figure 2 shows another kind of value function related to remediation (sanitation) time. It translates time measures

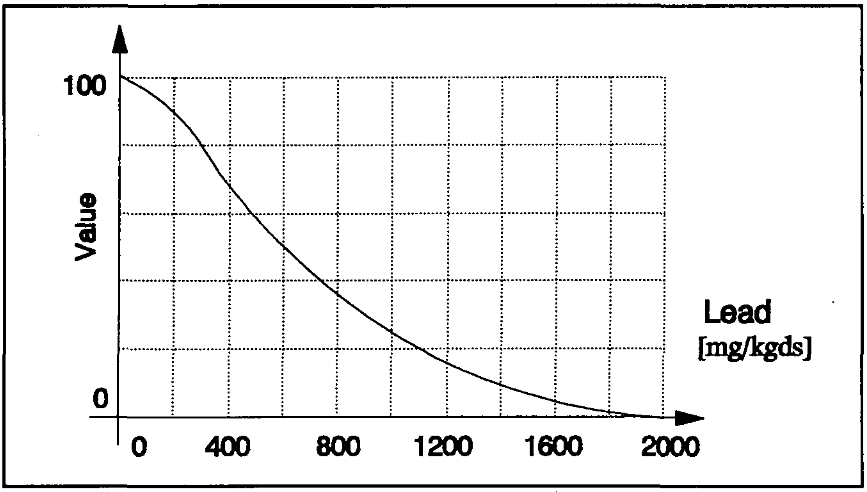

Figure 1. Value function for lead soil pollution.

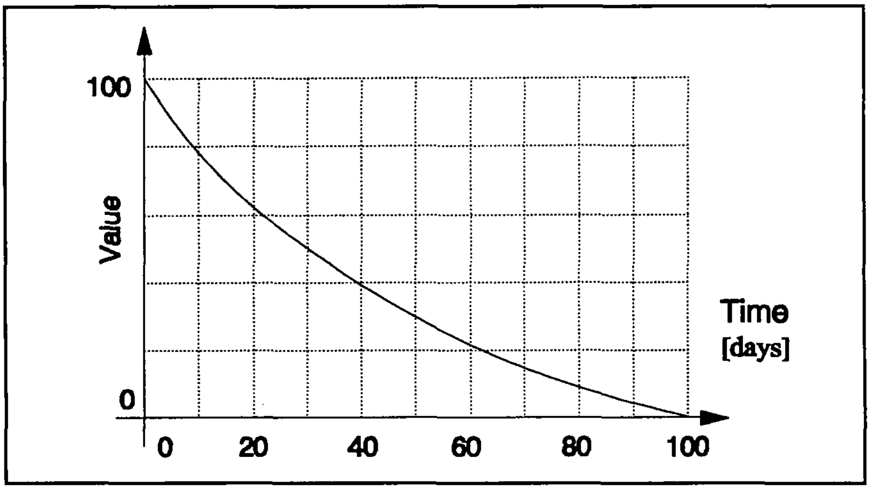

Figure 2. Value function for sanitation time.

into value scores representing the preference accorded to each possible clean-up duration.

The third step to build the value model refers to the weighting phase. Intuitively, weights represent the relative importance of one criterion against another. More precisely, "the relative weight assigned to an attribute should be proportional to . . . the change in overall value produced by moving the attribute from its least to its most valued state, all other things being equal" [12]. Weights are indicated as w_i, $i = 1,...,n$, and are normalized to add up to 1.

Finally, the fourth step is the additive combination of the individual value functions into an overall value function. The overall value score for the generic alternative $A = (x_1,...,x_n)$ is calculated as:

$$V(A) = V(x_1,...,x_n) = \sum_{i=1}^{n} w_i \cdot v_i(x_i) \qquad (1)$$

The model assigns to each alternative a score between 0 and 100, where the limit scores correspond to the limit performance profiles: $v(x_{1*},...,x_{n*}) = 0$, $v(x_1^*,...,x_n^*) = 100$. Given a set of alternatives, they can be ranked according to their value score and the cardinal rank order can be used to highlight the best alternative.

The additive aggregation of individual functions through weighting factors is justified under some precise conditions [13, 14]. Intuitively, this is related to independence of evaluation criteria, so as that each criterion score contributes independently to the overall value score of each alternative.

VALUE SCORES AND SOIL QUALITY

Multiple criteria additive value functions have been successfully applied in many applications related to environmental management [10, 15]. It is important to note that value scores can only be interpreted against reference situations and do not have an absolute meaning. This behavior originates from the characteristics of unidimensional value functions used for each criterion. In Figure 1, for instance, the concentration lead = 600 mg/kgds is assigned a value of fifty. This means that, provided the concentrations lead = 2000 mg/kgds and lead = 0 mg/kgds are given a precise value meaning, the concentration lead = 600 mg/kgds is half way down between these two situations, but does not mean that it is acceptable or unacceptable in absolute terms. Any absolute statement concerning values is meaningless and thus misleading.

In this article we are concerned with the elicitation of value functions and weights for the residual concentrations of contaminants after clean-up. This subproblem has some distinctive features which make the assessment particularly critical. Value functions for soil pollutants can be interpreted as substitutes for unavailable dose-effect functions and, more in general, as relative indices of soil quality. This introduces several issues, such as the relationship between the value functions, dose-effect functions and soil quality.

Furthermore, the model aggregates value functions for different pollutants through an additive role. This operation implicitly assumes a precise knowledge on effects aggregation for contaminant mixtures.

Soil Quality, Dose-effect Functions and Environmental Standards

Soil quality is an environmental index which relates soil pollutants to their effects. Environmental indices take either the form of indices of pollution or of indices of quality [16]. The difference is that as pollution increases, pollution indices increase while quality indices decrease. However, provided consistent units of measure are assumed the two indices are complementary and give the same information.

A soil quality index can be determined as a function of contaminant concentrations and of side conditions, such as soil type and soil usage. In Equation (2) soil quality (Sq) is a function of p contaminant concentrations ($c_1,\dots,c_p$) and of an additional factor Q[1] which encloses all other variables influencing soil quality.

$$Sq = f(c_1,\dots,c_p,Q) \tag{2}$$

[1] For the sake of the notational simplicity the factor Q will be omitted in the remaining text. Any consideration on soil quality measures, therefore, will be implicitly conditional to the known Q.

Soil quality may have several interpretations related, for instance, to the risk level for human or animals, to the percentage of protected species etc. By selecting a quality definition, the measurement scale and quality units are also specified. In some cases a scale from 0 to 100, for instance, can be appropriate.

Simpler expressions can be used if Sq can be determined combining quality functions for individual contaminants. The simplest case is the additive combination, as shown in Equation (3):

$$Sq = g(Sq_1(c_1),\ldots,Sq_p(c_p)) = \sum_{i=1}^{n} Sq_i(c_i) \tag{3}$$

where Sq_i, $i = 1,\ldots,n$, represents single-pollutant quality measures. The global Sq is a combination of the Sq_i through a combination rule g, which in Equation (3) is additive. Assuming Sq to be determined by the effects of contaminants, dose-effect functions are the basis of the Sq_i's and the function g would represent the effect-interaction mechanisms among pollutants.

"A damage, or dose-effect, function is the quantitative expression of a relationship between exposure to specific pollutants and the type and extent of the associated effect on target population" [16]. The estimation of a dose-effect function for a pollutant requires significant modeling and data collection. This is due to multiple exposure routes to the contamination, the transfer rates between external dose (indirect cause of effects) and internal dose (direct cause of effects), and the complexity of the models to estimate effects [17]. The complexity of the evaluation probably explains why "dose-effects relationships are simply unavailable for some 90% of the chemical agents produced on a commercial scale" [18].

On the basis of scientific and experimental evidence the Sq_i's in Equation (3) are not available. In addition, there is little knowledge on the interaction mechanisms for pollutant mixtures and little scientific proof supports the assumption that global effects of mixtures can be predicted from the effects of individual substances (cf. [19] for mixtures of metals in soil). Therefore, the models in Equations (1) and (2) have little applicability on the basis of laboratory data; however, they are useful to interpret the approach of standard concentrations in terms of soil quality.

Environmental standards are based on risk assessment procedures. To set a standard for a contaminant it is first necessary to fix a level of acceptable risk and then the corresponding trigger concentration which separates the acceptable and unacceptable levels of pollution [20]. Dose-effect functions are at the basis of this approach. In general, given a specific definition of environmental quality, a standard concentration for a contaminant is the concentration to which corresponds the lower acceptable quality, as shown in Equation (4):

$$c_i^{st} : Sq_i(c_i) = Sq_0 \tag{4}$$

where $c_{i}st$ is the standard concentration for the i-th substance. If Sq_i is strictly monotonical, there is a biunique correspondence between quality and concentrations: any statement about quality can be substituted by a statement about concentrations. When multiple contaminants are considered this does not hold and a quality level might correspond to many different combinations of contaminants.

Standards for multiple contaminants cannot, in general, be expressed in terms of concentrations for single contaminants, unless the combination of contaminants corresponds to non ambiguous situations (i.e., sufficient conditions). For instance, ideal soil quality might be assumed if all concentrations are below some strict levels for each pollutant, regardless to the actual single level. Similarly, concentrations exceeding a fixed level for one or more contaminants might unambiguously represent unacceptable quality. In symbols, in the first case there exists a combination $c^* = (c_1^*,...,c_p^*)$ such that if $c_i \leq c_i^*$ for every $i = 1,...,n$, then $Sq(c_1,...,c_p) \geq Sq^*$. In the second case there exist c_{i*}, $i = 1,...,n$, such that if $c_i \geq c_{i*}$ then $Sq(c_1,...,c_p^*) \leq Sq_*$ for $i \varepsilon \{1,...,n\}$, where Sq^* and Sq_* represent a high and low reference quality levels respectively.

A Decision Making Perspective of Soil Quality

Assuming a soil quality measure is available, Figure 3 shows the role of standard concentrations and of quality measures for a simple case with two hypothetical substances, Substance 1 (S_1) and Substance 2 (S_2).

The quality index maps combination (c_1,c_2) into quality numbers and the curves in Figure 3 show iso-quality combinations measured on a scale from 0 to 100. The straight lines corresponding to c_1^*, c_{1*}, c_2^* and c_{2*} represent standard concentrations for the substances, while the curve Sq_0 represents a quality standard.

In this example, the combination $c^* = (c_1^*,c_2^*)$ is the objective of clean-up operations. At the opposite end, concentrations c_{1*} and c_{2*} represent the maximum acceptable levels for individual contaminants. Therefore, any combination in the dashed area is certainly unacceptable while any combination in the cross-dashed area is certainly acceptable. Intermediate situations are determined through quality standards such as Sq_0; the dotted area indicates combinations which obey the standard.

The clean-up goal of the soil guideline is to reduce contaminants below the A-levels representing multifunctional soils: $c^* = (c_{1*},...,c_{p*}) = (A_1,...,A_p)$. Since this objective of unconditionally high soil quality is difficult to achieve it is necessary to compromise soil quality against other performances, such as clean-up costs. Figure 4 shows the residual concentrations for six hypothetical clean-up alternatives, $A_1,...,A_6$.

Alternative A_6 is discarded on the basis of standard concentrations while no available alternative meets the objective $c^* = (c_1^*,c_2^*)$. By fixing a quality standard equal to ninety, for instance, alternatives A_3, A_4 and A_5 are also discarded and the selection is limited at A_1 and A_2.

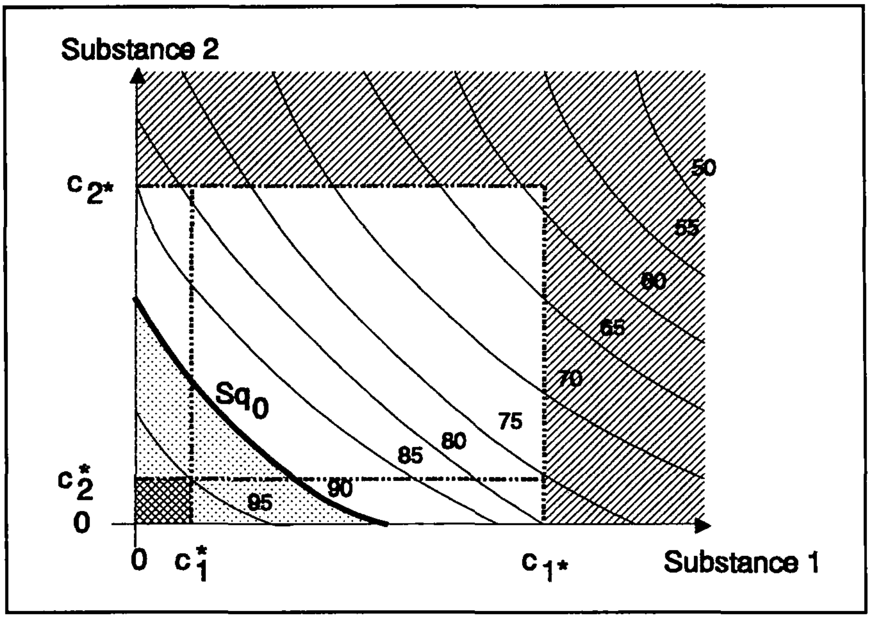

Figure 3. Concentration standards, iso-quality curves and quality standards for two pollutant substances.

This reasoning based on standards does not give much insight in the decision process. If A_3, for example, has a very low cost compared to A_1 and A_2 it might be convenient to accept a lower soil quality and select A_3 due to economic considerations. This approach is especially useful when soil sanitation is applied in real cases and it is necessary to accept compromise and sub-optimal solutions due to budget limits.

This has relevant consequences for the assessment of the soil quality index. Once lower bounds such as c_{1*} and c_{2*} and ideal levels such as c_1^* and c_2^* are fixed, quality measures are relevant only for a subset of concentrations within these bounds. The main objective of the quality index becomes that of quantifying the relative position of clean-up alternatives rather than that of measuring soil state in absolute terms. A relative quality index which maps mixtures of contaminants between the two limit combinations (c_1^*,c_2^*) and (c_{1*},c_{2*}) is equally suitable for this purpose.

Value Functions as Substitute Measures of Soil Quality

Soil quality indices are difficult to estimate due to the lack of sufficient laboratory data and effect studies. In this and similar cases it is a common practice

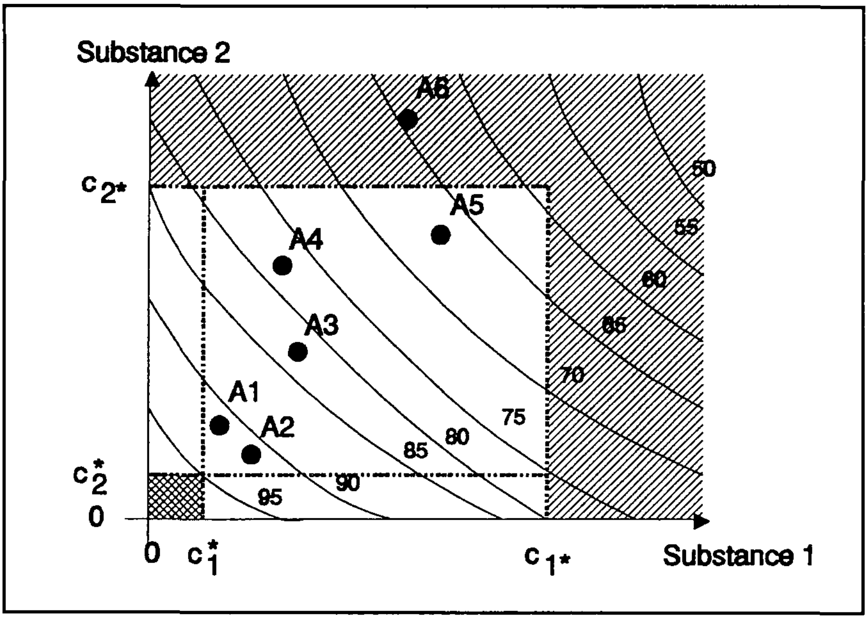

Figure 4. Residual contaminant concentrations for a set of six cleaning-up alternatives.

to refer to expert judgement when formalized estimations are unavailable [18]. On the basis of Equation (2), this would mean that experts evaluate the effects of pollutant mixtures and, for each possible level of contamination, estimate the corresponding soil quality level. Apart from very simple cases, this process is unrealistically difficult to be of practical relevance [21]. Therefore, the quality index has to be determined through Equation (3).

This approach is meaningful if the effects of a contaminant mixture can be determined combining effects of individual pollutants evaluated separately. Some studies are available on this subject [19] but a precise and definite answer is still lacking. Berg and Roels provide an evaluation scheme for simultaneous exposure based on expert judgement [22]. Their scheme is based on full or partial additivity. Full additivity is assumed for organic compounds while partial additivity, where global effects are lower than the sum of singular effects, is adopted for metals and inorganic compounds. In this scheme, brought into the Dutch legal guidelines, additivity plays the central role. If it is assumed for any situation it provides results which at maximum emphasize the actual effects, implicitly underscoring safety concerns.

Following all these considerations, there is a natural link between soil quality indices and additive value functions. Both associate a composite index to combination of pollutants and, provided additivity holds, determine a global index combining individual indices. In addition, in the clean-up framework only relative quality measures are necessary, and value function models strictly operate in this setting. Value function models have been explicitly designed to represent human preferences and judgements in a formal framework, and specific assessment techniques have been developed. Since laboratory data is missing, soil quality needs to be defined through expert judgements and value function techniques provide the necessary assessment tools. Therefore, in practical applications the value function model seems appropriate to substitute the unavailable indices of soil quality.

This can be formalized within the framework presented in Section 3. Let $x_* = (x_{1*},...,x_{p*})$ and $x^* = (x_1^*,...,x_p^*)$ be two limit combinations of p contaminants such that $Sq(x_*) < Sq(x^*)$. Let us also assume there exist p unidimensional value functions v_i associating pollutant concentrations to relative values representing individual soil quality indices v_i: $[x_i^*,x_{i*}] \rightarrow [0,100]$, i = 1,...,p. Finally, let the global value be a weighted additive combination of unidimensional values which maps contaminant mixtures into a value score such that $v(x_*) = 0$ and $v(x^*) = 100$:

$$V(x) = V(x_1,...,x_p) = \sum_{i=1}^{p} w_i \cdot v_i(x_i) \qquad (5)$$

for any $x = (x_1,...,x_p)$ such that $x_i^* \leq = x_i < x_{i*}$, i = 1,...,p. The use of value functions as substitute measures of soil quality is based on the following substitution:

$$v(x) = \frac{Sq(x) - Sq(x_*)}{Sq(x^*) - Sq(x_*)} \cdot 100 \qquad (6)$$

which simply states that within the set $\{x = (x_1,...,x_p): x_i^* \leq = x_i \leq = x_{i*}\}$ normalized value scores are used instead of unavailable quality scores.

A TECHNIQUE TO ASSESS VALUE FUNCTIONS FOR SOIL QUALITY

Several assessment techniques have been designed for evaluating v(x). Beinat reviews the major methods and highlights several shortcomings for their use in this particular domain [21]. The requirements for a good assessment technique can be summarized as follows. First, the procedure should allow qualitative and tentative responses. The value attached to a concentration synthesizes a complex analysis of effects and the experts find it difficult to give precise estimations. Similarly, although the weights have an intuitive meaning, the assessment of precise numerical weights is regarded as very controversial and the experts tend to avoid it. Since the value functions model requires numerical functions and weights, another requirement is that of estimating the model from a set of qualitative and imprecise judgements. Finally, the assessment procedure should be easy to use, interactive and flexible, so as to allow the experts to decide the course of action and avoid a "black-box" procedure.

The Assessment Technique

The description which follows focuses on the main points of a new assessment technique specifically designed for soil quality; the interested reader can find detailed description in [21] and [23].

Starting from unidimensional value functions, for each substance the experts indicate a range of possible values for a set of reference concentrations. This range is called value region and is likely to include the real curve. However, experts are not forced to give precise and unreliable estimations and they just indicate where the curve is likely to fall. Figure 5 shows an example of a value region for cadmium obtained by interpolating three value segments assessed for the A, B and C concentrations [1]. In a similar fashion, experts estimate a qualitative importance ranking for the substances, which corresponds to a qualitative estimation of weights. Value functions and weights estimated in this way constitute the direct assessment.

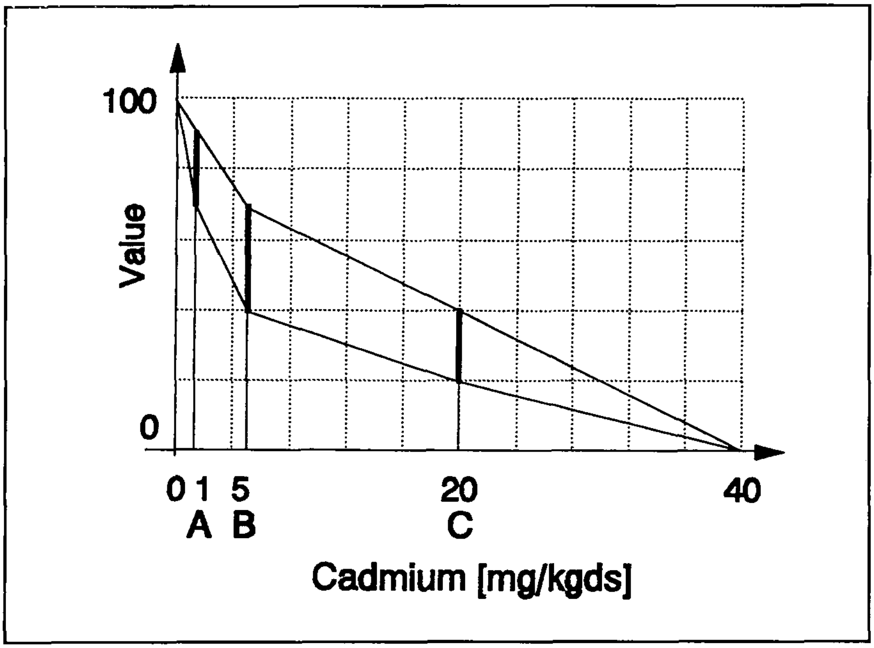

Figure 5. Cadmium value region.

This information represents what experts are willing to assess without being forced into unreliable numerical judgements. Since it is not sufficient to specify the value model it is necessary to gather further data from another perspective. This is the role of the indirect assessment. Experts are asked to assess simple combinations of contaminant concentrations and evaluate their clean-up priority on a qualitative scale. Figure 6 shows an example with nine combinations of cadmium and zinc. The numbers represent their clean-up priorities assessed through expert judgement. Number one indicates the highest priority and thus corresponds to the lowest soil quality.

The direct and indirect assessment essentially provide the same information from different perspectives. This results in a surplus of information which can be used to estimate precise functions and weights on the basis of simplified expert inputs. This is obtained using an optimization technique, such as a linear programming model. The idea is that of selecting the set of value functions and weights

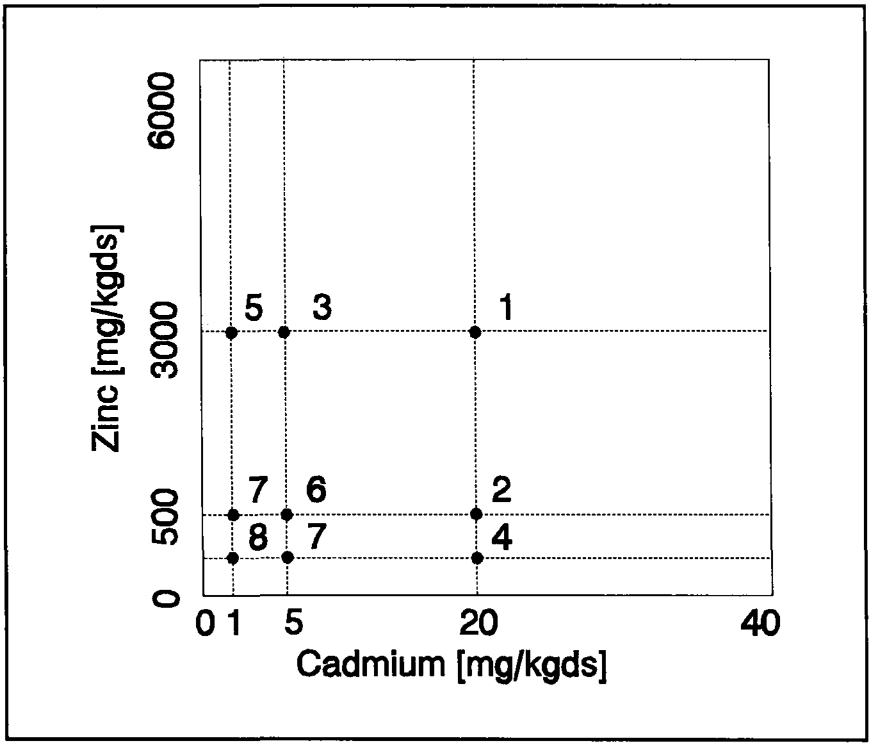

Figure 6. Indirect assessment for combinations of cadmium and zinc. Numbers represent cleaning-up urgency (1 = the highest) or soil quality (8 = the highest).

which: 1) obey the value regions; 2) obey the weight order; 3) evaluate combinations of contaminants in agreement with expert judgement. Since assessment inconsistencies are likely to occur, it is not always possible to achieve this result. In such a case the objective of the optimization procedure is that of estimating the set of functions and weights which maximize the consistency with all expert inputs.

A specific software system has been designed to assist the assessment. The software EValue [24] guides the experts toward the definition of the correct set of value functions and weights in a series of steps. The software has three basic editing modules, for value functions, weights and indirect assessments, an optimization module, which includes facilities to customize the optimization, and a system module, to analyze technicalities of the assessment, such as consistency levels. The block diagram in Figure 7 shows the structure of EValue.

The experts can specify data with various degrees of accuracy. Value regions, for instance, can range from very narrow or even precise curves to broad regions which carry virtually no information. These data are the input of an optimization module which, in real time, provides the corresponding set of numerical value functions and weights. Through the editing modules, they are presented to the experts and compared against the original estimations. If refinements are necessary due to high inconsistency of results or if experts do not consider the results as

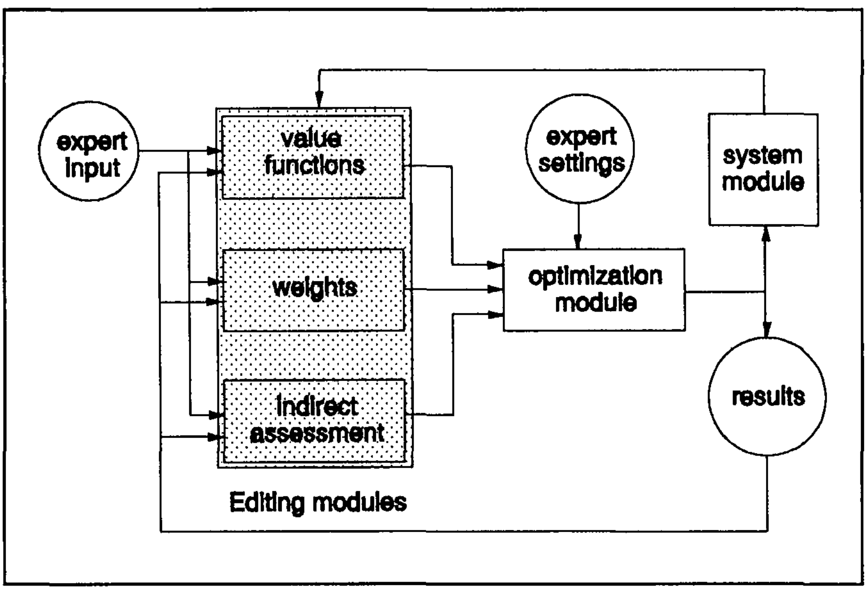

Figure 7. Block diagram of the software EValue.

satisfactory, editing of original assessments can take place so as to proceed to a second round. This process continues until satisfactory results are achieved.

APPLICATIONS

EValue has been applied in several assessments with many different experts. As an illustration, here follows the assessment of three value models. Every model has been estimated in five separate sessions with five well known experts of Dutch public organizations [24]. Before the assessment the experts received the same information on soil composition, usage and groundwater position, along with a short description of EValue.

The range of concentrations for each substance considered was always fixed from 0 to 2C mg/kgds (milligrams of pollutant per kilogram of dry soil weight), the C level representing the trigger for cleaning-up. This fixed the limit combinations for each case, x_i^* and x_{i*}. Table 2 summarizes the data for the three cases, along with the A, B and C concentrations for each pollutant.

For the indirect assessment it was necessary to select the pollutant combinations to evaluate. Given the second case in Table 2, for instance, twenty seven simple combinations of contaminants organized in three groups were used. The first group regarded the nine possible combinations of A, B and C levels for cadmium and zinc, keeping mineral oil at a zero level. Similarly, two other sets of nine combinations were defined keeping cadmium first and then zinc at the zero level.

Table 2. Assessment Cases: Concentrations are in Milligram of Pollutant per Kilogram of Dry-Soil Weight (mg/kgds)

		Reference Concentrations (mg/kgds)				
		x_i^*	A	B	C	x_{i*}
Case 1	lead	0	50	150	600	1200
	mineral oil	0	100	1000	5000	10000
Case 2	cadmium	0	1	5	20	40
	zinc	0	200	500	3000	6000
	mineral oil	0	100	1000	5000	10000
Case 3	cyanide	0	5	50	500	1000
	benzene	0	0.01	0.5	5	10
	PCAs	0	1	20	200	400
	chlorobenzene	0	0.05	1	10	20

Assessment Results

The complete assessment session for all substances took, on average, two hours per expert. The first case with two substances was almost always the longest because it also served to familiarize with the software and the assessment procedure. As the functioning was clear, the assessment took place in a much simpler and quicker way. For all sessions and for all experts it was necessary to run the procedure more than once. For the first case inconsistencies between assessments and results were always very limited after the first round. However, due to its simplicity the experts wanted to make changes in the assessments and to test the corresponding results.

The second and third case required more thorough analysis. As it is natural to expect, increasing the number of substances and the number of judgements required for the assessment makes it more and more difficult to obtain overall consistency. In general, however, the final results were either totally consistent with the assessments or showed very limited inconsistency [24].

Two samples of results for the second case are shown in Figures 8 and 9; complete results can be found in [24]. Figure 8 shows cadmium value functions

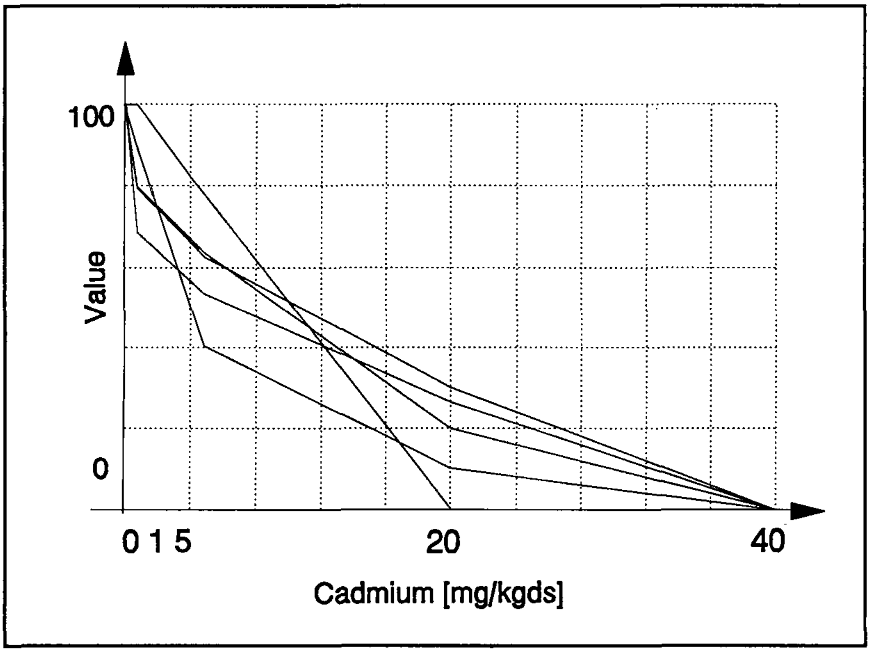

Figure 8. Value functions for cadmium of five experts.

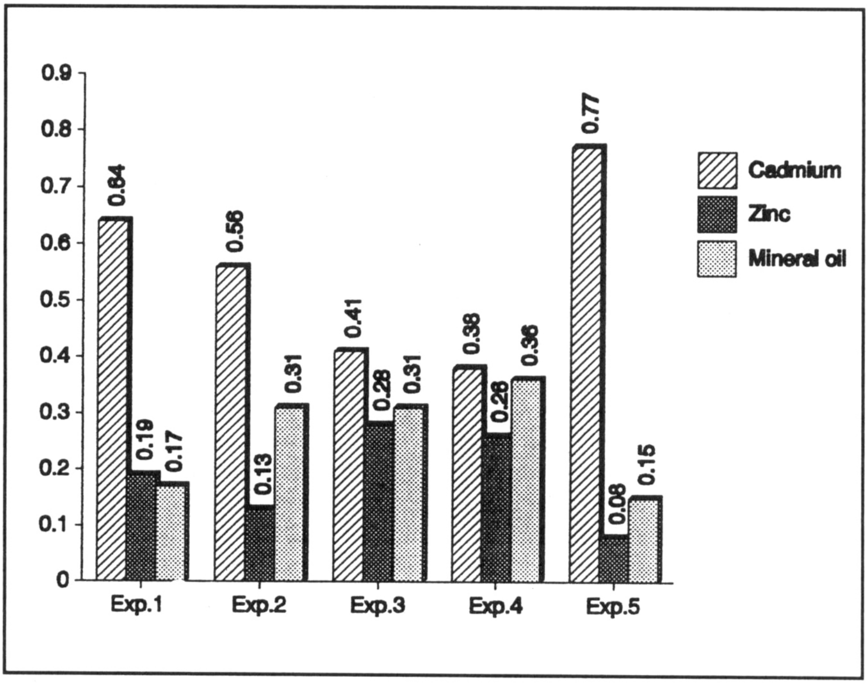

Figure 9. Weights for cadmium, zinc, and mineral oil of five experts.

for the five experts and Figure 9 shows the corresponding weights for cadmium, mineral oil, and zinc.

Analysis and Evaluation of Outcomes

As can be seen from Figure 8 and 9, value functions and weights can be rather different among experts. A common trait of the assessments was that during the various iterations the indirect judgements, i.e., the judgements on the clean-up priority of combinations of contaminants, were considered of primary importance. This means that the experts were rather strict in demanding value functions and weights which rank priorities, or soil quality, consistently with their indirect judgements, even if considerable changes were necessary for single value functions or weights.

The favorable attitude toward the indirect assessment was also the reason for a favorable evaluation of the whole assessment procedure and results. Before starting the sessions the experts were almost invariably skeptical, considering the elicitations of value functions and weights as difficult and unreliable. They also considered the process of substituting soil quality with a value model as an interesting idea, but of little practical relevance. The possibility of testing the results against judgements of clean-up priorities (soil quality) served as a test of the reliability of the assessments. By checking that value functions and weights correctly imitate expert responses in the test situations, experts positively considered their use in real situations and ended up considering this system an interesting practical way to evaluate clean-up outcomes.

The effect of this high importance of the indirect assessment is that value functions and weights are individually estimated essentially on the basis of qualitative considerations, such as a correct shape of the function, instead of on an analysis of numerical point values. This fact has several consequences. The first is that major attention is given to a correct final use of the model, i.e., to results which correctly emulate expert judgement and experience. The numerical interpretation of value functions and weights is difficult due to the fact that they are primarily functional to a correct evaluation of contaminant mixtures. This could be seen along the assessments when significant changes to the value function were accepted in order to respect the indirect priorities. However, the specific evaluation of single values attached to a contaminant concentration is considered as difficult regardless to any other consideration. The curves are rather analyzed against the expected behavior of the pollutant, such as limited or no effects for low concentrations, thresholds of effects and so on.

This fact introduces some complications into the analysis of the additive model. In the experiments the results always showed almost total adherence to the additive model, since the clean-up priorities calculated with the model and those indicated by the experts are almost totally consistent [24]. This can be explained by assuming that, according to the perception of experts, there is a natural additivity of effects. On the other hand, since experts accepted to adapt value functions and weights in order to represent correctly the clean-up priorities, it could simply be that the model is the best additive approximation of the combined effects, while the combination rule remains unknown. At this stage of the research, however, both solutions are possible and further analysis are needed to clarify this point.

CONCLUSIONS

This article proposes an alternative model to evaluate soil quality in the presence of contaminants and a new technique to estimate soil quality with multiple criteria value functions. The model demonstrated that value functions can be used as substitutes of unavailable soil quality measures in the evaluation of conflicting alternatives. Although value functions do not have the same degree of accuracy of dose-effect functions and cannot be interpreted as precise effect indicators, they can represent expert judgement and reproduce expert preferences with precision. These features are sufficient in a decision environment where trade offs between alternative solutions have to be made and the objective is that of highlighting the most advantageous compromise solution.

The assessment technique designed for this environment and the software EValue proved useful in real situations. The combination of direct and indirect assessment simplifies expert's task and responds to expert's attitude and ability to provide estimations of soil quality. Experts' confidence in the final results has always been high and experts favorably evaluated the use of value models as substitutes of quality measures in the decision making process.

The experiments showed that different experts tend to provide different responses and evaluate the same reality in different ways. At this stage of the research, the problem of aggregating expert responses and determining an "average expert" model remains still open. In spite of this, results for each expert demonstrated high self-consistency, indicating that the model is able to adapt to different perceptions and evaluations keeping the assessment to high levels of quality.

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