Stellingen

behorende bij het proefschrift van J.P. Okx, *Soil remediation. A systems approach*, Landbouwuniversiteit Wageningen, 9 november 1998

I

Teneinde te komen tot een efficiency- en effectiviteitsverbetering van het bodemonderzoeks- en bodemsaneringsproces zullen we - ondanks de rol die Peters voor een multidisciplinaire aanpak ziet weggelegd - de gangbare mono- en multidisciplinaire paden moeten verlaten en zullen we ons meer cross- of transdisciplinair moeten gaan bewegen.

Peters, T. (1992) *Liberation management.* Pan Books, London

Dit proefschrift

II

De stelling van Brus dat het een wijdverbreid misverstand is dat een statistische aanpak van bodembemonstering tot hogere kosten leidt, gaat geheel voorbij aan het utiliteitsvraagstuk en is dus in haar algemeenheid onjuist.

Brus, D.J. (1993) *Incorporating models of spatial variation in sampling strategies for soil.* Proefschrift Landbouwuniversiteit te Wageningen

Dit proefschrift

III

Informatie omtrent de onzekerheid van de verontreinigingssituatie verkregen door toepassing van geostatistische technieken dient te worden gebruikt bij de kostencalculatie van saneringen en vormt derhalve één van de mogelijkheden om het door Groen aangegeven handelen op basis van onzekerheden te operationaliseren.

Groen, M. (1998) *The IQ of the soil.* CUR/NOBIS, Gouda

Dit proefschrift

IV

Het genereren en kiezen van bodemsaneringsalternatieven dient te worden beschouwd als een *modified search decision process* of een *dynamic design decision process*, maar niet als een *basic search decision process*.

Janssen, R. (1991) *Multiobjective decision support for environmental problems.* Proefschrift Vrije Universiteit te Amsterdam

Dit proefschrift

V

De systeembenadering toegepast op het bodemonderzoeks- en bodemsaneringsproces leidt in tegenstelling tot hetgeen Hudson suggereert niet zozeer tot één algemeen geldend paradigma voor alle binnen het werkveld vertegenwoordigde disciplines, als wel tot een paradigmatisch raamwerk waaraan de paradigmata van de verschillende disciplines dienen te worden aangepast.

Hudson, B.D., 1992. The soil survey as paradigm-based science. *Soil Science Society of America Journal*, 56

Dit proefschrift

VI

Een geografisch informatie systeem kan de identificatie van verontreinigde deelgebieden aanzienlijk verbeteren en vergemakkelijken en vormt derhalve een ideaal startpunt voor een geautomatiseerde ruimtelijke *search* routine binnen de *development* fase van het bodemonderzoeks- en bodemsaneringsproces.

VII

In het geval dat verschillende actoren verschillende beslisregels hanteren is onderhandelen over de te nemen beslissing wellicht effectiever dan het trachten vast te stellen van één gezamenlijke beslisregel.

VIII

Divergeren alvorens te convergeren tijdens de *development* fase van het algemene besluitvormingsproces van Mintzberg is essentieel indien men de kennisbasis wil verbreden.

IX

Werken volgens de in de gangbare kwaliteitssystemen voorkomende normen leidt niet per definitie tot een product dat aan alle gestelde eisen en verwachtingen voldoet.

X

Onzekerheid is het substraat van de wetenschap.

XI

Niets is zo theoretisch als de ideale praktijk.

XII

Het nadere voorschrift bij artikel 5.1 van het promotie-reglement van de Landbouwuniversiteit Wageningen waarin staat dat volgens goed gebruik de promovendus iedere stelling die niet op het proefschrift betrekking heeft met de hoogleraar bespreekt tot wiens vakgebied het onderwerp van die stelling behoort geeft aan dat niet alle tijd van een hoogleraar nuttig dient te worden besteed.

SOIL REMEDIATION

A Systems Approach

Promotores: dr. ir. J. Bouma
hoogleraar in de bodeminventarisatie en landevaluatie
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bij het Departement Omgevingswetenschappen

SOIL REMEDIATION

A Systems Approach

PROEFSCHRIFT

ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van de Landbouwuniversiteit Wageningen
dr. C.M. Karssen,
in het openbaar te verdedigen
op maandag 9 november 1998
des namiddags te vier uur in de Aula

door

Johannes Pieter Okx

Cover painting: *A distant mirror* [detail], Oil on paper 128 × 96 cm, Jan Schut (1998)

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De spatten en vlekken die ernaast zitten zijn eigenlijk de wonden,
de cicatrices van het gevecht met de materie.

Karel Appel in de documentaire film "If I were a bird" (1994) van Mat van Hensbergen

Voor Heleen, Thom en Lisette

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Voorwoord

Bij de totstandkoming van dit proefschrift kon ik terugvallen op de kennis en kunde zoals aanwezig binnen de Sectie Bodemkunde en Geologie van het Departement Omgevingsvraagstukken van de Landbouwuniversiteit Wageningen, het Wagenings Instituut voor Milieu- en Klimaatstudies en Tauw Milieu. Het is goed om de bodemsaneringspraktijk van nabij te kennen teneinde er iets zinnigs over te kunnen schrijven, maar het is nog beter je zo af en toe aan de praktijk te kunnen onttrekken om een wat objectiever oordeel over die praktijk te kunnen vellen en eventuele alternatieven te bedenken. De momenten dat ik mij aan de praktijk heb kunnen onttrekken en mij volledig aan dit proefschrift kon wijden heb ik derhalve als bijzonder positief en verrijkend ervaren. De ivoren toren is dan ook wat mij betreft – weliswaar als wisselwoning – aan herwaardering toe!

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Abstract

Okx, J.P., 1998. **Soil remediation. A systems approach.** Doctoral thesis, Wageningen Agricultural University, The Netherlands.

Soil remediation has only a short history, but the problem addressed is a significant one. When solving soil remediation problems we have to deal with a large number of scientific disciplines, however solutions are often presented from the viewpoint of just one discipline. In order to benefit from the combined disciplinary knowledge and experience it is necessary to describe the interrelations between these disciplines. This has been realised by developing an adequate model of the desired process, which enables to consider and evaluate the essential factors as interdependent components.

Three main phases in the soil remediation process are distinguished: problem identification, development of problem solving alternatives and selection of the best alternative.
In the identification phase several sampling strategies may have to be compared. In this thesis probabilistic decision trees are used for the comparison. In the case studies we found that the value of surveys depends not only on the costs of the survey itself, but equally on the ratio of expected failure or success and the related costs of the actions based on the survey. Once a sampling strategy is chosen and data is collected, the results can be used to estimate the amount of polluted soil material. Probability kriging is a non-linear geostatistical estimation technique suitable for the estimation of the amount of polluted soil material.
In the development phase work is aimed at generating problem solving alternatives. This thesis presents expert support models recombining knowledge and experiences obtained during ex and in situ soil remediations. The aim of the models is to optimise knowledge transfer among the various parties involved in contaminated site management. Structured Knowledge Engineering (SKE) has been used as a framework for model development. The model was applied several polluted sites. The structured approach requires scrutinising all relevant data in order to answer the questions related to ex and in situ soil remediations. Moreover, it clarifies the roles of the different disciplines involved in the process.
After deciding whether or not a soil cleanup operation is necessary, the question remains which remedial strategy and technique should be applied. The triple-perspective REC framework simultaneously takes into account risk reduction, environmental performance and costs, and aims at increasing the effectiveness and efficiency of cleanup operations.

Additional index words: soil remediation, systems science, decision making, ex situ soil remediation, in situ soil remediation, expert support system, geostatistics

Chapter 1

INTRODUCTION

Sokrates: Herinnert ge u niet dat ge gezegd hebt, dat het vak van verteller een ander is dan dat van wagenmenner?
Ioon: Ja, dat weet ik nog.
Sokrates: En omdat het een ander vak is, vindt ge dan met mij dat het ook kennis van andere zaken inhoudt?
Ioon: Ja.
Sokrates: Dus kan het vak van verteller volgens u niet alle kennis bevatten, noch kan de verteller zelf alles weten.
Ioon: Behalve misschien de genoemde voorbeelden, Sokrates. [1]

1.1 General

Concerns arose about what we inherited from previous generations when it became apparent that a number of residential and recreational areas had been built on heavily contaminated soil or chemical waste (Otten et al., 1997). Events such as the Japanese "Hexavalent Chromium Incident" in Tokyo in 1975 (Gotoh and Udoguchi, 1993) or the Dutch Lekkerkerk case in 1980 brought about the development of soil protection policies with the intention to protect public health and environment against adverse effects of soil contamination. These policies resulted in numerous remedial actions.

The number of contaminated sites is enormous and so are the costs. In Europe and the United States the clean-up operation is expected to cost more than 1,400 billion ECU. Whatever the exact figures will be, such an enormous operation deserves serious attention. Wrong estimations may possibly lead to political and social turbulence and clean-up operations will inevitably be slowed down.

Soil-water-air environments are extremely complex and different soil fractions and constituents give rise to diverse reactions when anthropogenic chemicals are introduced (Cairney and Hobson, 1998), thus making correct estimations is far from an easy task.

---

[1] Taken from Platoon (428/427 – 348/347 BC), "Ioon"; [Dutch translation], De Driehoek, Amsterdam

The toxicity of contaminants in soil is associated with soil properties that affect the bioavailability of the contaminants, such as pH, cation exchange capacity (CEC) and composition and concentration of soil organic matter. In soil ecosystems, these parameters vary as a function of time and space (Marinussen, 1997). In many cases, contaminants become bonded, particularly to clay and organic content particles, and so are unavailable to present future risk. In other situations introduced chemicals may remain unbonded, or can be remobilised by changes in soil acidity or redox potential, and then become far more able to create risk (Cairney and Hobson, 1998). As most of the corrective remedial actions nowadays are risk-based corrective actions, it is obvious that knowledge of the above-mentioned processes and their spatial distribution is essential for risk assessment.
Soil properties, however, are also to be evaluated when deciding on the remedial actions for most of these actions are based on trying to change the availability of the introduced chemicals.

Many scientists and engineers agree that problems concerning soil remediation require at least an interdisciplinary approach. In the private as well as in the public sector, however, responsibilities are more and more decentralised. Therefore knowledge about the business process is dispersed. As a consequence multi- or even monodisciplinary approaches are still very common and knowledge is often acquired through analysis rather than through synthesis. As a consequence, soil remediation lacks an explanatory concept providing a foundation and structure for scientific research as well as for the environmental production sector performing the clean-up operations. As a result soil remediation research as well as clean-up operations will fail to develop into optimal processes.

This image of today's soil remediation research leads to the aim of this thesis that may be summarised as: to foster soil remediation research towards a fully-fledged problem-oriented discipline in order to yield efficient and effective solutions for soil pollution problems. Two core objectives are derived from this single aim:

- to supply soil remediation research with a explanatory concept or a paradigmatic framework to guide the future research;
- to facilitate consistent problem analysis and decision making.

The following sections will focus on what is needed to foster soil remediation research towards a fully-fledged problem-oriented discipline.

1.2 Environmental science

Environmental science theory

Soil remediation research could benefit from the recent advances of environmental science. Environmental science as a problem-oriented discipline belongs to a subgroup of the normative family of sciences. Although the boundaries are vague, the normative sciences consist of three types of disciplines (De Groot, 1992):

(1) Ethics, occupying itself with general values and normative procedures.
(2) Problem-oriented disciplines, focussing on areas of societal problems. Compared to ethics they are much more concrete and 'filled with facts'.
(3) Design-oriented disciplines, differing from the problem-oriented ones in that they are grounded more in generalised societal demands than in concrete problems: civil engineering in the generalised demand for efficient infrastructure, agricultural science in the generalised demand for secure food production, and so on.

Design is an inherent element in the problem-oriented disciplines, but these designs arise as answers (proposed solutions) to concrete questions (problems). In the design-oriented disciplines, the designs predominate.

The situation regarding soil remediation research now is comparable to that of environmental science a few years ago. Environmental science education has been characterised by rapid developments in the early nineties. Up to that time, only minor subjects and field research had been supplied to neighbouring, monodisciplinary departments. In those days, although students could study environmental science for two years, they remained students in biology, sociology or some other discipline. Environmental science had a tradition of 'problem hopping' without much reflection on general methodologies or on the normative principles that are applied to define what is a problem or a good solution at all (De Groot, 1992). However, today environmental science is a fully-fledged problem-oriented discipline. Soil remediation research, however, has failed to benefit from the recent advances of environmental science. Practitioners study soil remediation problems, but remain soil scientists, geologists, hydrologists, microbiologists or whatever.

Common to all sciences is a notion of methodological circularity. In the positive branch of empirical science there is the 'empirical cycle': hypotheses are deduced from general theories, they are tested in real-world cases, and the results are fed back into the theory. Usually, three steps are distinguished. The first is characterised by terms such as problem identification, problem description, problem diagnosis, problem analysis, modelling and so on. The second step is characterised by terms such as design, policy formulation, plan evaluation and so on. The third step is usually called implementation (De Groot, 1992).

Systems approach

Despite the notion of methodological circularity, suggesting some kind of system, classical science has far more concern with thinghood than systemhood. In fact, the many disciplines and specialisations that have evolved in science during the last four centuries reflect predominantly the differences between things rather studied than the differences in their way of being organised (Klir, 1991). Drilling equipment, chemical analytical instruments, shovels or bioreactors are well within the competence of the engineer trained in the respective discipline. But when it comes to solve environmental problems such as the greenhouse effect or the soil pollution problem, the call for interdisciplinary teams is often heard. However, experience has emphasised that this is not a successful way to tackle such problems, rather the fact that it is quite difficult for specialists from one discipline to understand the concepts and language of another (Checkland, 1976).

We need cross- or transdisciplinary concepts which serve to unify knowledge by being applicable in areas which cut across trenches that mark traditional academic boundaries. Systems science provides such concepts, for (Klir, 1991):

- Systems science and methodology are direct applicable in virtually all disciplines of classical science;
- Systems science has the flexibility to study systemhood properties of systems and the associated problems that include aspects derived from any number of different disciplines and specialisation's of classical science. Such cross- or transdisciplinary systems and problems can thus be studied as wholes rather than collection of the disciplinary subsystems and subproblems;
- The cross- or transdisciplinary orientation of systems science has a unifying influence on classical science, increasingly fractured into countless numbers of narrow specialisation's, by offering unifying principles that transient its self imposed boundaries.

The application of the cross- or transdisciplinary concept to the problems related to soil remediation allows us study the problem as a whole and not as a collection of unrelated subsystems and subproblems. This does not mean that disciplines become invisible. In fact a certain discipline may seem to be dominant in a particular stage of the soil remediation process. However, the discipline will always be subordinated to the problem.

1.3 Decision processes

Many descriptions of the concept *problem* can be found in the literature. According to Monhemius (1984) an individual has a problem if he finds himself in a situation in which he experiences a discrepancy between his notion of the desired reality and his perception of the reality, and wishes to eliminate that discrepancy. According to Ackoff (1981) a problem is a situation that satisfies three conditions: first, a decision-maker has alternative courses of action available; second, the choice made can have a significant effect; third, the decision-maker has some doubt as to which alternative should be selected. In this thesis we will use Ackoff's definition of a problem.
A general model for *problem solving* or *decision making* is given by Mintzberg et al. (1976). They distinguish three main phases of decision-making: problem **identification, development** of problem solving alternatives and **selection** of the best alternative (Figure 1.1). Ex situ and in situ remediation design involves identification and development of problem solving alternatives. The phases can easily be identified in most guidelines for contaminated soils.
The **identification** phase consists of the central routines: *recognition*, in which the problem is recognised and evokes decisional activity and *diagnosis*, in which the decision makers seek to comprehend the evoking stimuli and determine the cause-effect relations for the decision situation.

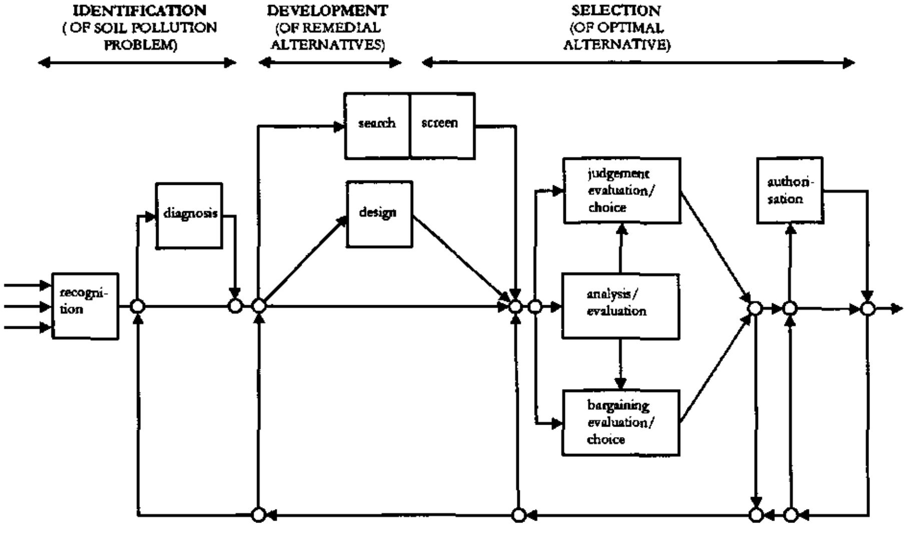

Figure 1.1. A general model of decision processes (Mintzberg et al., 1976)

The **development** phase contains a *search* routine to find ready-made solutions and a *design* routine to develop tailor-made solutions. Finally, the **selection** phase consists of a *screen* routine, several *valuation/choice* routines and an *authorisation/implementation* routine. Thus, Mintzberg's model serves the 'empirical cycle' and it describes the way our 'soil remediation system' works.
Thus, the benefits of a systems approach are demonstrated in three critical phases in the decision process:

1. Decisions on identification or characterisation of the soil pollution problem (see Chapters 3 and 4);
2. Decisions on development of ex and in situ soil remediation concepts (see Chapter 5 and 6);
3. Decisions regarding the selection of a suitable solution for the addressed problem (see Chapter 7).

1.4 Spatial statistics

As discussed in the previous section we can deal with generating possible solutions for a soil pollution problem, only after the problem is described. In this thesis I will restrict myself to spatial description, which is usually called geostatistics. Geostatistics are the application of the theory of the regionalised variables (Matheron, 1960) to the estimation of all kind of deposits. More generally, when a phenomenon spreads in space and exhibits a certain spatial structure, we shall say that it is regionalised (Matheron, 1971). In its simplest form, a geostatistical model can be written as:

$$z_i = \mu + \varepsilon_i$$

where z_i is the value of Z at any location i, μ is the mean of Z and ε_i is a spatially correlated random component whose variation is defined by a semivariogram.
Early applications of geostatistics were in mining. An author like Krige (1951) will be forever linked to ore evaluation. Much later the theory found its way to soil science. Early applications in soil science are to be found in Burgess et al. (1980) and Webster (1980). Stein (1991) offers not only an overview of spatial interpolation, but in addition gives many examples of applications of geostatistics in soil science. Finally, soil pollution problems are addressed by Leenaers et al. (1988), Okx et al. (1992), Boekhold (1992) and many others. The geostatistical descriptions are frequently used to estimate the volume of polluted soil that is one of the key cost factors in soil remediation.

1.5 Outline of this thesis

This thesis is a compilation of articles published in or submitted to scientific journals but subsequently slightly modified.
Chapter 2 introduces soil remediation in its present form. The first part gives an overview of the extent of the problem. The second part points out the major problems related to soil remediation operations. The chapter ends with giving a number of suggestions to handle these problems, to be worked out in the rest of the thesis.
Chapter 3 deals with decision theory as a tool for the valuation of investigation strategies. It is shown that a combination of decision trees and probability assessment tools such as statistics and geostatistics are useful for a priori evaluation of chosen strategies. It also shows the importance of feedforward and feedback mechanisms in achieving the considered goals.
Chapter 4 shows two applications of probability kriging. This technique can be used as a probability assessment tool in the above-mentioned valuation process as well as a tool to provide 3D-models of the polluted subsurface, which are used as the basis for the design of remedial alternatives. The technique is applied to two cases. The first case describes a heavy metal pollution caused by atmospheric deposition stemming from a zinc factory in the south of The Netherlands. The second case addresses a heavy metal pollution related to a former cotton mill in the city of Haarlem in The Netherlands. The chapter emphasizes the importance of describing uncertainties regarding the estimated volume of contaminated soil for cost estimations.
In Chapter 5 an expert support model for the design of ex situ soil remediation alternatives is presented. It recombines knowledge and experiences in order to optimize knowledge transfer among the various parties involved in contaminated site management and remedial design. Structured Knowledge Engineering (SKE) has been used as a framework for model development. The model was applied to a hydrocarbon pollution at a former fuel station as well as to a polycyclic aromatic hydrocarbon pollution at a former gasworks. It is shown that the model provides the answers to the most relevant questions regarding ex situ remediation.
Chapter 6 presents the in situ version of an expert support model. The model requires scrutinizing of all relevant data to answer questions related to the design of in situ soil remediation alternatives. The model was applied to a chlorinated hydrocarbon pollution at a dry cleaner's. It is shown that the model supplies answers to the questions necessary to make decisions regarding in situ soil remediation.

After a number of remedial alternatives are worked out, the question remains which remedial strategy or techniques should be applied. In the first part of Chapter 7 the triple-perspective REC framework that enables to answer this question is presented. The framework simultaneously takes into account risk reduction, environmental performance and costs, and aims at increasing the effectiveness and efficiency of cleanup operations. In the second part the REC framework is applied to chlorinated hydrocarbon pollution caused by a dry-cleaner. Finally, Chapter 8 offers an overview of the results in view of the objectives of this thesis as well as some thoughts on possible future developments.

Chapter 2

THE NATURE OF SOIL REMEDIATION PROBLEMS

Soil remediation has only a short history, but the problem addressed is a significant one. Cost estimates for the clean-up of contaminated sites in the European Union and the United States are in the order of magnitude of 1,400 billion ECU. Such an enormous operation deserves the best management it can get. Reliable cost estimations per contaminated site are an important prerequisite. In this thesis we will address the problems related to site-wise estimations.

When solving soil remediation problems we have to deal with a large number of scientific disciplines. Too often solutions are presented from the viewpoint of just one discipline. In order to benefit from the combined disciplinary knowledge and experience we think that it is necessary to describe the interrelations between these disciplines. This can be realised by developing an adequate model of the desired process, which enables to consider and evaluate the essential factors as interdependent components of the total system.

The resulting model provides a binding paradigm to the contributing disciplines that will result in improved efficiency and effectivity of the decision and the cost estimation process.

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Not to see the forest for the trees is a serious failing. But it is an equally serious failing not to see the trees for the forest. One can only plant and cut down individual trees. Yet the forest is the "ecology", the environment without which individual trees would never grow. To make knowledge productive, we will have to learn to see both forest and tree. We will have to learn to connect. [2]

2.1 Introduction

Background

The interest in the field of soil remediation from all parts of society has been considerable over the past decade and has resulted in the rapid development of the environmental production sector. What started as the subject of a few environmental activists is now an important source of employment. In general, the sixties, seventies and eighties have been characterised by an increasing awareness for environmental problems related to water, air and soil, respectively (Carrera and Robertiello, 1992). Hence, soil remediation has only a short history.

In 1980 the Unites States federal government promulgated the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) normally addressed as the Superfund Act. Initially approximately 30,000 sites had been identified, but nowadays it is believed that about 75,000 sites may benefit from the Superfund Act. Moreover, within the realm of the more recent Resource, Conservation and Recovery Act (1976), it is believed that remedial action is necessary for another 37,000 sites (Russell, 1991). The cost of the entire US operation is expected to be in excess of 880 billion ECU.

In the various EU member states the number of contaminated sites was estimated to be approximately 55,000 (Merzagora, 1991). Total cost of the initial long-term clean-up programs of the European Union member states was estimated to be in the order of 100 billion ECU (Porta, 1991). However, as indicated in the proceedings of the international workshop on contaminated sites in the EU (Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, 1995), the 1994 predictions of the various member states sum up to a total of over 500,000 sites (see Table 2.1).

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[2] Taken from Peter Drucker , "Post-Capitalist Society" (1993), Harper Business, New York

Table 2.1. Inventory of contaminated sites in the EU (Anonymous, 1994)

Country	Number of sites	Source
Austria	24,155	1)
Belgium	9,000	2)
Denmark	10,000	1)
France	667	3)
Germany	143,252	1)
Greece	-	
Ireland	-	
Italy	9,805	1)
Luxemburg	-	
Netherlands	200,000	1)
Portugal	-	
Spain	4,532	1)
Sweden	1,700	1)
United Kingdom	100,000	1)
total EU	502,444	

1) Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, 1995. Proceedings of the International Workshop on Contaminated Sites in the European Union, December 1994, Bonn.
2) OVAM, 1990. Verontreinigde sites. Voorbereiding ontwerpplan 1991-1995. OVAM, Openbare Afvalstoffenmaatschappij voor het Vlaamse Gewest.
3) Ministère de l' Environnement, France.

Moreover, as the following Dutch example shows, early estimations could be underestimations. In 1981, as a result of the so-called Lekkerkerk affair (Vijgen, 1992), the then Ministry of Public Health and Environmental Hygiene, now Ministry of Housing, Physical Planning and Environment (VROM), executed the so-called Ginjaar inventory. In this inventory, all provinces reported the number of hazardous waste dumps. This first estimate yielded about 4000 suspect locations, about 1200 of which required further study. Finally, about 350 of these locations were found to qualify for immediate remediation. Cost estimates made in 1981 concerning the entire operation amounted to approximately 460 million ECU. This inventory and the subsequent cost calculations were repeated in 1984 as well as in 1986 and new estimates of respectively 920 and 1380 million ECU were published. In the mid-eighties, it became apparent that also current industrial sites were contaminated (Eikelboom and von Meijenfeldt, 1985; Gravesteyn, 1990; Holtkamp and Gravesteyn, 1993). The present estimate of the number of contaminated sites ranges between 195,000 and 315,000 and the current cost estimate is about 38 billion ECU (Vijgen, 1992).

One of the conclusions in the report "Soil remediation" of the Dutch Auditor's Office (1993) was, however, that the initial estimated average cost estimate for larger sites of 0.46 million ECU per case turned out to be close to 1.15 million ECU. This could mean that the present estimate of 38 billion ECU will have to be adjusted in the future.

Extrapolation of the Dutch findings to the EU - which is a dubious procedure and only yields very rough estimates - provides an estimate of the number of contaminated sites ranging between 500,000 and 700,000 and a cost estimate which could be between 225 billion and 800 billion ECU.

Whatever the exact figures will be, such an enormous operation deserves all the attention it can get. Faulty (under)estimations will lead to political and social turbulence and the clean-up operations will inevitably be slowed down. Both the number of cases and the cost per case are difficult to estimate. In this article we will discuss possible causes and we will suggest some solutions to improve the estimations of the cost per case.

To discover the problems linked to estimation of the costs of soil remediation we will first describe the nature of soil remediation. The description focuses on the types of problems as encountered in the soil remediation process. A description of the problems will be given in the next section. Finally we will give some suggestions to improve the estimations.

2.2 A complex and turbulent business

In soil remediation one has to deal with a large number of scientific disciplines: physical geography, soil science, geohydrology, biology, ecology, toxicology, biotechnology, chemistry, chemical technology, civil engineering, geostatistics, sociology, psychology, law and economics. The full extent of the relations between the disciplines is not fully known, but a first attempt to show these relations as concerns the soil remediation process is given in Figure 2.1.
Despite these relations, many of the existing problems within the process are studied separately from each other. Such analysis also tends to get increasingly profound and consequently a synthesis becomes increasingly difficult.

Moreover a considerable number of social viewpoints or perspectives may have to be taken into account while making decisions. Viewpoints of policy makers, polluters and house tenants regarding a polluted area may differ drastically from each other. The process of solving soil contamination problems has been described as a situation in which a large number of relevant perspectives has to be taken into account. These perspectives are also characterized by a high level of complexity.

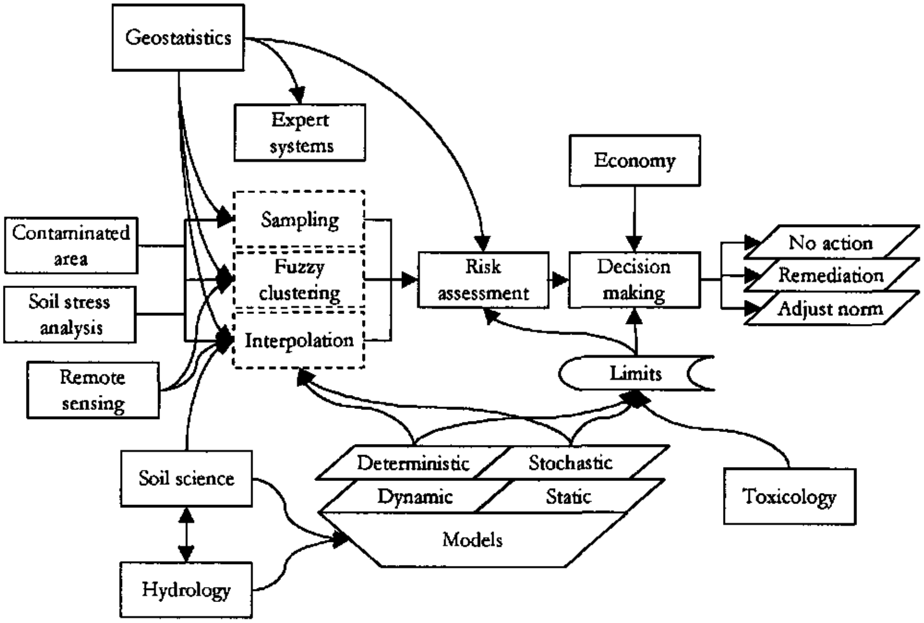

Figure 2.1. A general scheme showing data processing within an interactive GIS applied to spatial variability of risk assessment for soil contamination research (Stein et al., 1994)

Complexity is not the only problem present day the soil remediation business has to face. Our society is also characterized by turbulence (Naisbitt, 1982). New technologies and the effective use thereof will shorten the time span for research and development, yielding shorter product life cycles. The same phenomenon is seen in policy making. In soil remediation, this is evident from the growing number of publications and regulations.

Since adaptability is a necessity for organizations (Ackoff and Emery, 1972) the described complexity and turbulence will influence the way in which organizations design their strategies and business processes.

2.3 Knowledge dispersion

Mono-, multi-, inter-, trans- or crossdisciplinary approaches can be distinguished in scientific activities. A monodisciplinary approach uses the knowledge and experience from one single discipline. Multidisciplinary work is created when a number of fields cooperate but the "borders" between each of them remain. An interdisciplinary approach can create a new dimension to merge the original disciplines through cooperation (Derks, 1977). The differences between these three approaches are illustrated in Figure 2.2.

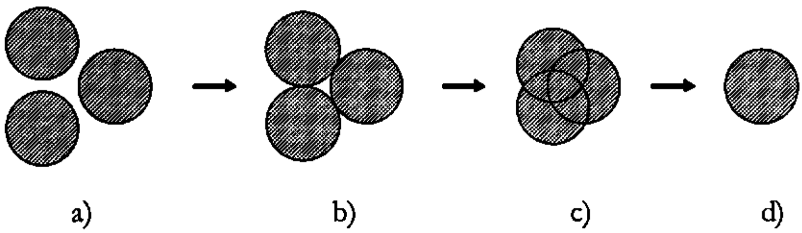

Figure 2.2. The development from mono-(a), to multi-(b), to inter-(c), to monodisciplinarity (d) (Derks, 1977)

However, the trans- or crossdisciplinary approach is not about merging different disciplines into a new holistic discipline, but rather about looking after the interrelations between these disciplines. Where the interdisciplinary approach inevitably leads to a decreased richness compared to the original disciplines, the trans- or crossdisciplinary approach insures the preservation of the richness of the originals.

Many scientists and engineers agree that problems concerning soil remediation require at least an interdisciplinary approach (Verkuijlen, 1989; Salomons and Förstner, 1988). In the private as well as in the public sector, however, responsibilities are more and more decentralised. Therefore knowledge about the business process is dispersed and partial for organisation members. As a consequence multi- or even monodisciplinary approaches are still very common and knowledge is often acquired through analysis rather than through synthesis, implying that the emphasis is on technical rather than on organisational issues.

As a consequence, the soil remediation business lacks a co-ordinating explanatory concept providing a foundation and structure for scientific research. Such an explanatory concept, or paradigm as it is called by Kuhn (1962, 1970) is necessary to make substantial progress. In absence of a paradigm, all facts that could pertain to an area of study are likely to appear equally relevant. As a result, scientific practice tends to be a nearly random activity (Hudson, 1992) in which only slow progress is achieved.

For the complex and turbulent remediation business to keep up with existing knowledge and remaining compatible within the environment means having to operate on a more abstract level and having to make strategic choices in knowledge acquisition for some common purpose. This is not easy for an individual, but it becomes even more difficult when an organisation as a whole has to make out the direction into which it wants to develop or strengthen its position. In the following chapter different viewpoints on how to acquire this knowledge will be discussed.

2.4 Achieving organisational effectiveness

General

Possible solutions for the optimisation of the effectiveness of organisations dealing with soil remediation can be found in management science, information science and systems sciences. However none of these sciences gives complete solutions.
Hammer (1993) - one of the pre-eminent management gurus of the 1990s - promotes a radical redesign of processes, organisation and culture to achieve a quantum leap in performance, but fails to say how this should be achieved.
Although a number of decision support tools have emerged from information science, important items in decision making like personal knowledge, intimate understanding of the business and "Fingerspitzengefühl" cannot be formalised into systems. Therefore, the knowledge that can be feasibly encoded in a system is not sufficient for decision-making (Luconi et al., 1984). Bots and Sol (1988) argue that an optimisation of organisational effectiveness can be achieved by paying attention to co-ordination of different actions for problem solving.
Finally systems science provide methods applicable to nearly all disciplines of classical science. It also has a unifying effect on the disciplines which contribute to a process (Klir, 1990).

The management science point of view

Where Senge (1990) argues correctly that traditional hierarchical organisations are not designed to provide people's higher-order needs, self-respect and self-actualisation and the importance of personal mastery, Hammer and Champy (1993) and Mintzberg (1973, 1994) manage to give an explanation for this phenomenon.

Hammer and Champy (1993) argue that the majority of organisations are built around Adam Smith's (1776) brilliant discovery that industrial work should be broken down into its simplest and most basic tasks. This practice resulted in a spectacular rise of productivity, but it also ended craftsmanship (Mintzberg, 1973) and personal mastery (Senge, 1990). In Smithsonian organisations, where responsibilities are decentralised, people simply became detached from their work. In the automobile-industry, where Henry Ford and Alfred Sloan (General Motors) developed division of labour into nothing less than a fine art, Volvo was probably the first to recognise the detachment of workers and a consequent loss of organisational effectiveness. As soon as Volvo reinvented car building by abandoning flow production or rather by reunifying tasks into one coherent business process, people became involved again. This way people's attitude was changed by changing the business process.
As we can see in Figure 2.1, soil remediation is a process in which the decentralisation of responsibilities seems almost logical. In Denmark sampling and interpretation are a responsibility of the local authorities, the decision on the remedial action, however, is taken by the national authorities. This results in low sampling and interpretation budgets, which are serious threats to the quality of the final decision making. The Danish Environmental Protection Agency therefore considers this division of labour and responsibilities as the main problem of the Danish soil remediation program. In the Netherlands the Minister of Housing, Physical Planning and Environment (VROM) also mentioned the decentralisation in the Dutch Soil Remediation Operation process as one of the causes of an insufficient control and progress of the operation (Dutch Auditor's Office, 1993).

Hammer and Champy (1993) believe that reunifying tasks into coherent business processes is essential for founding and building of corporations in the post-industrial business age we are about to enter. They stress the importance of looking at the process and argue that one way to do so is through 'business reengineering'. Business reengineering is the fundamental rethinking and radical redesign of business processes to achieve dramatic improvements in critical, contemporary measures of performance, such as cost, quality, service, and speed. It is not about making incremental changes that leave basic structures intact (Hammer and Champy, 1993). The same reunification of tasks is an important prerequisite for changing people's attitude, because reunification of tasks generally leads to a deeper understanding of the business. This, in turn leads to a situation in which workers feel that they make a difference! It is this situation which will make people feel responsible and motivated to challenge the goals of soil remediation and they will therefore willingly contribute to its success.

Although professionals and managers agree that business reengineering is an important tool, differences in attitude towards the organisation between both groups are a complicating factor. Professionals tend to take their own decisions without too much attention for the organisational structures, whereas managers are far more loyal to their organisation. Where professionals find restructuring of the business as proposed by Hammer and Champy (1993) a prerequisite for changing people's attitude, managers consider it a direct threat to the stability of the organisation and are reluctant to discuss the matter. Nevertheless, business re-engineering happens to be a possible solution for problems concerning communication, attitude and performance and should therefore be discussed.

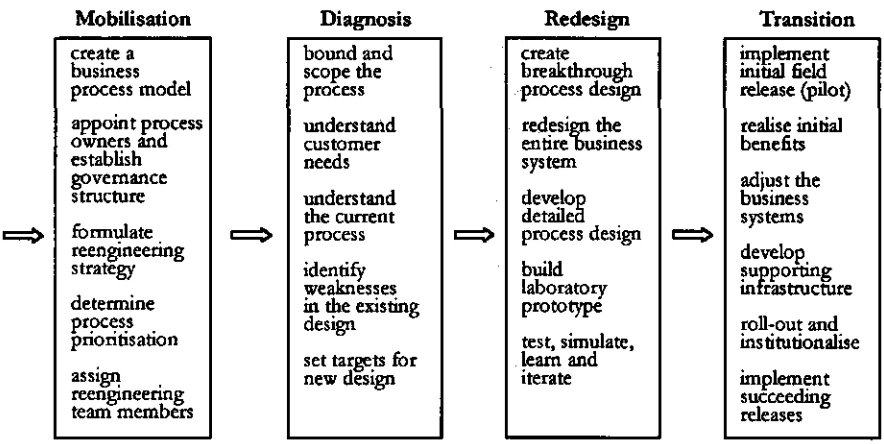

Figure 2.3. Procedure for business reengineering (Hammer, 1994)

Business reengineering is an interesting concept, but unfortunately procedures given for reengineering are not very detailed (see Figure 2.3). Creating a breakthrough process design concept, redesigning the entire business system, developing a detailed process design sounds very reasonable, but the question remains: how to do it?

The information science point of view

Dearden (1972) as well as In 't Veld (1988) reported that managers have been thinking in processes and systems for centuries and that this way of thinking is quite useful. In the early sixties many organisations entered the era of computer-based information systems, which makes information technology relatively new in the management business. In the sixties Transaction Processing Systems (TPS) and

Management Information Systems (MIS) emerged (van Weelderen, 1991). Both system types were aimed at collecting, updating and presenting information according to predefined procedures. The MIS was aimed at providing aggregated information to the management. Mintzberg (1973) however considered oral media, hearsay, gossip, tangible details and speculation as far more important. In the early seventies Gorry and Scott Morton (1971) introduced the Decision Support System (DSS) which was aimed at improving the quality of decisions. These DSS's were mainly concerned with organisational problem solving (van Weelderen, 1991). In the late seventies the Knowledge Based System (KBS) or Expert System (ES) concept emerged. According to Zeidner et al. (1986) this concept is aimed at replacing the engineering experts with software that emulates their behaviour and rationale. Decision making, however, goes beyond the processing of well-structured intellectual knowledge, analytical reports, abstracted facts and figures. Personal knowledge, intimate understanding of the business and "Fingerspitzengefühl" should be considered equally important and this kind of knowledge cannot be formalised into a KBS or ES. Luconi et al. (1984) consequently argued that the knowledge that can be feasibly encoded in an ES is not sufficient to make decisions by itself and introduced the Expert Support System (ESS) concept. The ESS concept is aimed at aiding, rather than replacing, the human decision-makers. The fact that decision making goes beyond formal knowledge was not acknowledged by Janssen et al. (1990) and - although their assessment procedure itself is elegant - this is probably the reason that their decision support system for management of polluted soils never found its way to the potential users.

The systems science point of view

Somewhere in between information science and systems science Bots and Sol (1988) made a distinction into three different perspectives from which organisational performance can be reviewed: the micro perspective, the meso perspective and the macro perspective. In the micro perspective is concerned with the workplace of the individual information worker in the organisation, and improvements at this level aim for an increased performance of such an individual (Sprague, 1986). The meso perspective is concerned with the business processes within an organisation, and improvements at this level are expressed in terms of specific characteristics of the product being made. As a consequence, from the meso perspective the focus is on the co-ordination of different information workers active within the same business process (Dur, 1992). The macro perspective is concerned with the common objective of an interorganisational system and its performance in achieving this objective. In this context boundaries between two or more organisations are disregarded (van Weelderen, 1991).

This way of thinking has a strong resemblance to the systems science way of thinking. In 'General System Theory', von Bertalanffy (1968) pointed out that the history of systems science dates back to the 15th century in which Nicholas of Cusa wrote his 'De ludo globi' (1463). However it took about five centuries before the subject became fashionable, as von Bertalanffy puts it. In 1967 the Canadian Prime Minister Manning wrote the systems approach in his political platform saying that '... an interrelationship exists between all elements and constituents of society. The essential factors in public problems, issues, policies, and programs must always be considered and evaluated as interdependent components of a total system'. In the literature dealing with general system theory, one finds wide divergence in the definition of systems, criteria of classification, and in the evaluation of the systemic approach as a contribution to knowledge, understanding, or pursuit of specific practical goals (Rapoport, 1986).

Klir (1990) gives a guided tour through systems science. He distinguishes three classes of problem-solving activities:

a. Systems inquiry. The set of activities to create a system that is an adequate model of the actual business process. These activities are the realisation of Hammer's mobilisation stage (see Figure 2.3).
b. Systems design. The set of activities to create a system that is an adequate model of the desired business process. This set is the concrete form to Hammer's diagnosis and redesign stages (see Figure 2.3).
c. Systems implementation. The set of activities to implement the designed system. This set models Hammer's transition stage (see Figure 2.3).

Building a system can be done in many different ways and indeed the established scientific disciplines all developed different preferred ways to divide the world into environment and system. These ways are strongly related to the paradigms mentioned earlier in this study.

Systems science, however, investigates the isomorphy of concepts, laws, and models from various fields and helps in useful transfers from one field to another. It also promotes the development of adequate theoretical models in fields that lack them. Moreover systems science minimises the duplication of theoretical effort in different fields by promoting the unity of science through improving communication among specialists (Klir, 1990). It is for this reason that systems science should play its role in the optimisation of the soil remediation process.

2.5 Towards an operational system

As we have shown knowledge on the process of remedial investigations is being dispersed and hardly anyone has a complete overview. In order to improve this unsatisfying situation we made a start to collect, examine and categorise all existing knowledge and experiences in the field of in situ remediations. The aim of the operational system is to optimise the transfer of knowledge and experience among the various parties involved in remedial investigations aimed at the design of in situ soil remediation.

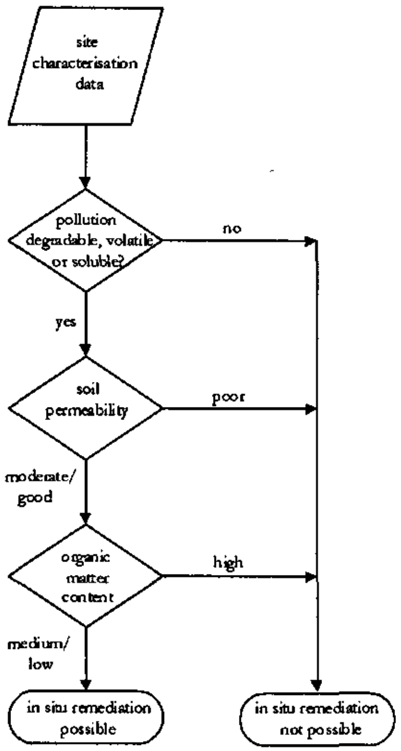

Figure 2.4. Process diagram of the feasibility of in situ treatment

Knowledge technology comprises a large number of methods that can be used in support of the development of expert support systems (ESS). In this example Structured Knowledge Engineering (SKE) (Bolesian, 1991) was used for the development of the system. SKE distinguishes four different phases: a preliminary phase, a feasibility study, analysis and the design. In both the analysis and the design phase, the information needed was obtained by means of literature research and interviews.

Table 2.1. Contaminant information on in situ treatment

Contamination	Volatility	Biodegradability		Solubility	In situ Possibilities
		Aerobic	Anaerobic		
Hydrocarbons					
Gasoline (C_4 - C_{12})	+	+	-	+	yes
Kerosene (C_6 - C_{15})	±	+	-	+	yes
Gasoil (C_9 - C_{26})	-	±	-	- *	yes
Domestic fuel (C_9 - C_{24})	-	±	-	- *	yes
Lubricants (C_{15} - C_{40})	-	-	-	-	no
Aromatics (BTEX)	+	+	±	+	yes
PAH					
Light (2-3 rings)	±	+	-	± *	yes
Heavy (4-5 rings)	-	-	-	- *	no
Chlorinated Hydrocarbons					
Aliphatic (per, tri)	+	-	+	+	yes
Chlorobenzene	+	+	-	+	yes
Pesticides	-	±	-	-	no
PCB	-	-	-	-	no
Heavy metals	-	-	-	± *	yes

* Solubility can be enhanced by detergents (for hydrocarbons) or by acidification (heavy metals)

In Figure 2.4 an example of a process diagram used in the system is displayed. It shows three simple questions which lead to the answer of the question "Is in situ treatment feasible?". The three questions can be answered by using Tables 2.1 to 2.3. The Tables are filled with state of the art knowledge and experience on in situ remediations. Table 2.1 answers the question whether contaminants prohibit in situ remediation. Table 2.2 gives an answer whether the soil type is a limiting factor for in situ remediation, just as Table 2.3 answers the question whether the organic matter content is a limiting factor. This structured approach forces the user to answer all the crucial questions needed for an in situ soil remediation operation and moreover it clarifies the position of the different disciplines in the process. The rest of the process diagrams concerning in situ remediation are shown in Chapter 6.

Table 2.2. Geohydrological information on in situ treatment

Soil type	K-factor (m/day)	In situ possibilities
Gravel	> 100	Yes
Very coarse sand	10 – 100	Yes
Coarse sand	5 – 10	Yes
Fine sand	0.2 – 5	Yes
Loam	< 0.2	No
Clay	< 0.2	No
Peat	< 0.2	No

Table 2.3. Information related to organic matter content

Organic matter content in %	In situ possibilities
0 - 1 low	Yes
1 - 5 medium	adsorption can give problems
> 5 high	No

2.6 Summary and conclusions

Discrepancies between cost estimations and cost realisations should be avoided. Organisations dealing with case-wise cost estimations must produce more reliable figures. To enable these organisations to do so it is necessary to provide a supporting system. This should be an adequate model of the desired business process, which enables to consider and evaluate the essential factors as interdependent components of the total system. Decision making differs from processing of well-structured intellectual knowledge, analytical reports, abstracted facts and figures, since it also involves personal knowledge and experience, intimate understanding of the business and a touch of Fingerspitzengefühl. Therefore we need a concept aimed at aiding, rather than replacing, the human decision-makers.

For this purpose, we identified three problem-solving activities: systems inquiry, systems design and systems implementation. In addition the perspective from which the problem is viewed is important. In soil remediation studies a strong preference exists for the analytical point of view, which is comparable to the micro perspective. Although there is no doubt that such tools are useful, we will devote ourselves to a more synthetic point of view, corresponding to the meso and the macro perspective. Many difficulties in trying to yield reliable estimates stem from the lack of a single binding paradigm. Systems science is able to provide such a paradigm. On the other hand an overaccentuation of the meso and macro perspective may withhold from the analysis what is needed for the synthesis. In other words the three perspectives can not be regarded in isolation from each other.

As soon as the desired business process is known, fallacies of the current system will become visible and a strategy of change can be elaborated. There are at least three parties that will benefit: the scientific community, the environmental production sector and the responsible authorities. In the scientific community the required interdisciplinary approach is often obstructed by the existence of too many separate disciplines within the (academic) institutions. Since interrelations between contributing disciplines are essential in systems design, the necessity of co-operation is obvious. System science is the most important tool to show how disciplines interrelate and should therefore be part of any scientific curriculum. The environmental production sector will benefit because their estimates will be far more accurate than before which should give them a competitive advantage. Finally the responsible authorities will be able to produce better figures for future planning.

Chapter 3

USE OF DECISION TREES TO VALUE INVESTIGATION STRATEGIES FOR SOIL POLLUTION PROBLEMS

Remediation of a contaminated site usually requires costly actions. Several clean-up and sampling strategies may have to be compared. In this chapter several common environmental pollution problems have been addressed by using probabilistic decision trees. Decisions in this chapter are on how to sample to detect a hot spot at different consumer's risks, whether to take additional samples to obtain a sufficiently precise estimate of an environmental contaminant, how to value environmental sampling strategies and how to choose between different test-procedures prior to the remediation of soils. Decision trees combine costs with possible actions and chance events. For the studies analysed they prove to be of use to make a well-supported decision. In the case studies we found that the value of surveys depends not only on the costs of the survey itself, but equally on the ratio of expected failure or success and the related costs of the actions based on the survey. Thus, minimising costs on surveys and tests can lead to considerable losses in terms of effectiveness and efficiency.

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Environmetrics: J.P. Okx, A.Stein

Art is a lie that lets us recognise the truth ... [3]

3.1 Introduction

A wide range of soil pollution problems is currently being addressed. Each pollution problem requires skilful treatment to avoid exceeding the estimated costs. Decisions therefore have to be made, for example on optimal sampling to discover hot spots, to estimate the average concentration or the amount of polluted soil, but also on balancing the choice of new *in situ* clean-up technologies against the traditional ones. Based upon collected data and intuitive knowledge on effectiveness, a remediation plan is made, including a list of options for different strategies (Okx et al., subm.). Also, during remediation itself decisions are to be made, like investing in large or small sums of money to collect additional observations, to collect the most informative set of data and to give reliable estimate of the size of the environmental problem to address. To the remediation plan costs are associated based upon economic estimates. At present no publications are available that deal with a balanced, quantitative way to select the best scenario.

Since we want to spend our money as effective as possible to solve as many problems as possible, it is important to value clean-up strategies in a realistic way. Each strategy, however, is subject to random events, like the probability to hit (or to miss) a hot spot, the spatial uncertainty caused by limited available data and effectiveness of a remediation technology. Therefore strategies consist of combinations of actions, random events and associated costs. So far, valuation depends upon best possible guesses from the literature, combined with a small number of observations (Raiffa and Schlaiffer, 1961; Raiffa, 1968; Baird, 1989; Clemen, 1995). Using statistical decision theory we may quantify the value of strategies (Berger, 1985; Puterman, 1994; Ten Berge and Stein, 1997). For example, soil surveys and field experiments should be realistic given the estimated costs and the costs of consequential actions: as data are expensive, they should be carried out as efficiently and effectively as possible to honour environmental objectives.

The objective of this study is to investigate how statistical decision theory can support to make a decision between different investigation alternatives. We focus upon four cases with objectives typical for soil remediation studies and taken from current studies at an environmental engineering agency.

[3] After Pablo Picasso (1881-1973)

3.2 Four cases

Case 1: A company wishes to take over a possibly contaminated 6 ha industrial complex. The source of possible contamination was a large shoe factory, which operated at the complex since 1946. The main environmental concern is presence of chromium VI, a degradation-product of chromium-sulphate, being used in the tanning-process. In 1991 the shoe factory moved its activities to another site. Taking over the site includes taking over liabilities for the contamination. Thus, estimation of clean-up costs prior to the take-over is desired. Although some original plans and aerial photographs were available, they proved to be of little value when trying to find the hot spots and a survey is used instead. The company has to choose between two survey schemes. Missing hot spots with a radius of 12.5 m will result in unforeseen remediation costs of approximately 90 kECU. The first survey scheme aims for total certainty: the chance of missing such a hot spot (the consumer's risk β) is equal to 0. The second survey scheme accepts a β value of 0.1. In this study, it is relevant to question whether investing in a more expensive scheme is advantageous to the company.

Case 2: This case considers a residential area of 1 ha polluted with lead in the Netherlands. Contamination sources were several small lead smelters operating within this area. They emitted large amounts of lead containing particles. The objective of the survey is to estimate the mean lead concentration and to check whether it is above an intervention value. The Dutch intervention value of lead is based upon clay and organic matter content, I_{Pb} = 6.24 A mg·kg^{-1} where $A = 50 + L + H$ mg·kg^{-1}, A being the target value, L the clay content (%) and H the organic matter content (%) (Van den Berg et al., 1993). Measuring instruments are standard. An initial sample of 20 observations is available. Exceeding the I_{Pb} would require an additional investment of 20 kECU, whereas a single observation costs 0.1 kECU. The question arises whether additional samples should be taken to make a reasonable save remediation decision.

Case 3: The third case study considers a residential quarter near the centre of one of the older Dutch cities, covering an area of approximately 17 ha. In the beginning of this century a cotton-mill was established in the eastern part of the district. The energy for the mill was provided by the factory's own gasworks. Discovering paint residues in pavements and gardens initiated a soil pollution research focussing on cadmium. The volume of contaminated soil must be estimated as precisely as possible because the total volume of polluted soil to be removed is the key factor in the cost calculations.

In the cotton-mill case the threshold, above which soil has to be removed, equals 5.0 $mg.kg^{-1}$ Cd. A common approach is:

1. Use the average of the calculations as the base scenario;
2. Produce a remedial action plan on the base case, and;
3. Use the probability of obtaining the base case and decide how much flexibility will have to be incorporated into the cost calculations.

The first two steps are quite common; the third step, however, a sensitivity analysis is seldom made. The costs (v_i) of a remedial alternative i are best represented by their expected value (E_i), to which the standard deviation (s_i) is added, multiplied by a risk avoidance factor (k) (Okx, 1998).

$$v_i = E_i + k \, . \, s_i$$

Case 4: In the fourth study a company plans to remediate a contaminated site. There are two alternatives: an expensive standard excavate and *ex situ* treatment of the excavated soil (500 kECU), and a relatively cheap innovative *in situ* treatment (250 kECU). If *in situ* fails, the *ex situ* technique has to be applied as well. On similar sites the *in situ* technique had a success rate of about 0.5. Before a decision for either alternative is made a laboratory column test (10 kECU) can be done. From experience it is known that in cases were the test indicated that in situ remediation is possible the success rate increased to 0.6. If the test indicates that in situ treatment is not possible the success rate decreased to 0.2. As an alternative, installing a small pilot plant at a cost of 75 kECU might be a better solution. From the past, it is known that the success rate equals 0.9 for C = 1 and 0.05 for C = 0. The question arises whether the column test or the pilot test should be performed.

3.3 Methods

A general structure for decision making

In this section we describe a general set-up for surveys and laboratory or pilot experiments for soil pollution problems. There are several principal steps that define an investigation strategy. We will distinguish between objectives, population, sample, degree of precision, measuring methods, sampling strategy and visualisation (Figure 3.1).

The objective of case 1 is to determine the possible presence of chromium VI (Cr-VI) hot spots, that is those locations where the concentration exceeds a pre-set threshold I_{Cr}. The objective function $\varphi_1(S,\beta)$, depending upon the sampling scheme S and the risk β, equals the costs to detect that [Cr-VI] > I_{Cr}. The objective of case 2 is to estimate an average lead concentration as precisely as possible. The objective function $\varphi_2(S,\alpha)$, depending upon the sampling scheme S and the risk α, equals the costs to make a reasonably sound remediation. The objective of case 3 is to estimate the volume of polluted soil to be removed from the area. Let the volume be given by V and let the set of possible contaminants be denoted by Z. Then it has been shown (Staritsky et al., 1992) that a good estimate is given by

$$\hat{V} = \sum_j I_Z(x_j)$$

where $I_Z(x_j)$ is the indicator function, taking the value 1 if any contaminant from the set Z exceeds the threshold t_z and 0 otherwise, whereas summation extends over the discretised volume of soil. The objective function $\varphi_3(S, \gamma, t_z)$, depending upon the sampling scheme S, variogram γ and threshold t_z , equals the minimised kriging variance.

The objective of case 4 is to find out which test procedures lead to the most efficient and effective remedial alternative. The objective function $\varphi_4(T)$, depending upon a choice for technology T, equals the total expected costs and we aim to minimise these.

A clear and unambiguous definition of the population must allow during a survey to decide whether a particular location belongs to it. The population of case 1 is an industrial complex of 6 ha. The depth to which the survey should extend is defined as the unsaturated zone. Similarly, the population of case 2 is the surface of the 1 ha residential area and of case 3 the first 4 m of the 17 ha residential quarter. The population of case 4 is the volume of soil to be excavated or to be treated.

To avoid large costs, all sampled environmental variables must be relevant to the objective of the survey, whereas no variables are omitted. For each of the cases 1, 2 and 3 historical information on shoe making, lead distribution and gaswork processing determine the variables to be measured. Also, clay and organic matter

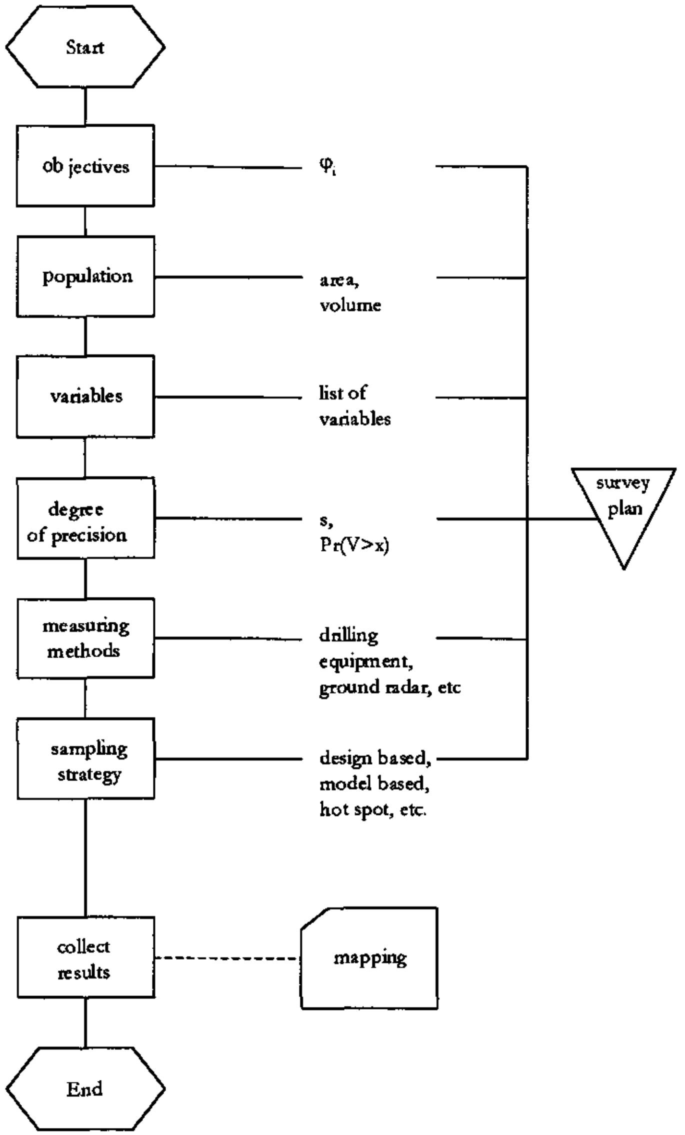

Figure 3.1. The general decision strategy for soil pollution problems

have to be measured to calculate the intervention level. For case 4, relevant data are related to the proposed *in situ* technique. The data must give a clear indication whether the *in situ* process occurs as predicted.

An essential element of statistical decision support is the precision to be reached in terms of the objective function. We distinguish between spatial uncertainty and non-spatial uncertainty. Spatial uncertainty is caused by the necessarily limited number of observations and can be reduced by taking more samples. Non-spatial uncertainty such as measurement error can be reduced by repeating the number of experiments or by improved measurement devices. Both are costly, and optimisation should obviously reflect the degree of required precision.

Measuring instruments need to be specified in relation to the objectives of the survey. Commonly arrangements have to be made with laboratories to have these available at the right time. In the first three cases no ambiguity exists, and well-defined methods can be applied. The fourth case, however, requires careful reflection. There are many ways and little standards to set up column experiments or pilot experiments. When designing such experiments the main question is whether the experiments yield the wanted data.

Strategies rely upon an effective sampling scheme indicating how many observations have to be taken and at which locations. Domburg (1994) distinguishes two objectives. The first objective is to estimate *how much* of a contaminant is present, leading to classical or design-based sampling (Cochran, 1977; Särndal, 1992). The second objective is to predict as accurately as possible *where* the contaminant is present, leading to geostatistical or model-based sampling (e.g. Webster, 1985; Van Groenigen et al., subm.). Design-based sampling yields unbiased estimation of the frequency distribution of the contaminants. Model-based sampling yields a description of spatial dependence and optimal interpolation at unvisited sites (Cressie, 1991). To these two we add the objective to identify the presence of hot spots. In that case we applied Singer's approach (Singer 1975, see appendix).

Finally, since decisions are to be made on the basis of the investigation, a clear presentation and visualisation of results is required. In particular geographical information systems can play here a central role. These show the extent of the environmental problem in its real context, they allow to make a link with hydrological models, show possible alternative strategies. For making the proper statistically based decision, the amount of error and uncertainty in the objective function can be presented as well.

Evaluating the value of soil surveys and experiments

The four different cases can be analysed using statistical decision trees. In a decision tree, squares represent decisions to be made, while the circles represent chance events. The branches emanating from a square correspond to the choices available to the decision-maker, and the branches from a circle represent the possible outcomes of a chance event. The third decision element, the consequence, is specified at the ends of the branches (Clemen, 1996). In decision trees both actions taken by the decision maker and random events are displayed, and associated costs and probabilities are quantified (Figure 3.2).

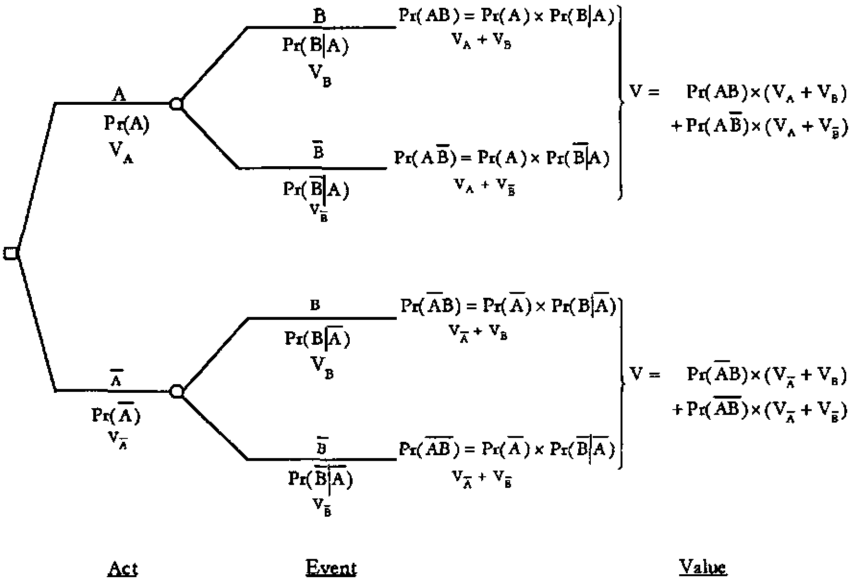

Figure 3.2. The statistical decision tree, including events (A,B), costs (C_i) and probabilities (Pr) and values (V). Squares indicate decisions, circles indicate chance events.

Suppose that at some stage a decision T_i among several alternatives T_k is made, $i = 1,\ldots,k$. Each decision is followed by a random event where A or $\overline{A}$ may occur with costs C_A and $C_{\overline{A}}$ and probabilities $Pr_i(A)$ and $Pr_i(\overline{A})$, respectively. This event is followed by a second random event where B or $\overline{B}$ may occur with costs C_B and $C_{\overline{B}}$, and conditional probabilities $Pr_i(B \mid A)$, $Pr_i(\overline{B} \mid A)$, $Pr_i(B \mid \overline{A})$ and $Pr_i(\overline{B} \mid \overline{A})$, respectively. The scheme can be extended in a straightforward way to include more random events as well, but we only consider two levels to make the presentation as transparent as possible.

In case of two events a standard Bayesian approach yields that

$$Pr_i(A|B) = \frac{Pr_i(AB)}{Pr_i(B)} = \frac{Pr_i(AB)}{Pr_i(AB) + Pr_i(\overline{A}B)} = \frac{Pr_i(A)Pr_i(B|A)}{Pr_i(A)Pr_i(B|A) + Pr_i(\overline{A})Pr_i(B|\overline{A})}$$

The joint probabilities can be obtained from decision trees. The value V_i for T_i, expressed as expected costs is then obtained as

$$V_i = C_{0,i} + C_A \,.\, Pr(A|B) + C_{\overline{A}} \,.\, Pr_i(\overline{A}|B)$$

where $C_{0,i}$ is an investment for strategy i, such as collecting an additional set of n observations, or investing into a laboratory test, C_A are the costs in case of event A and in case of A. The best decision is the decision with the lowest expected costs. Therefore, based upon possible decisions, the aim is to calculate the expected loss and to choose the decision with minimal expected loss.

As an example, a company considers to invest in taking over a possibly contaminated site. Two clean-up alternatives are considered: a relatively cheap innovative technology T_1 and a more expensive proven technology T_2. Failing risks for T_1 are higher than those for T_2. A choice for T_1 with costs C_1 may save considerable money. On the other hand, if T_1 fails, the problem remains unsolved and T_2 with costs C_2 is applied. The decision tree (Figure 3.3) shows that for T_1 a chance event (success or failure) determines the final costs. The chance of success, i.e. ending with the lowest costs, is compared with the risk of ending up with the highest costs.

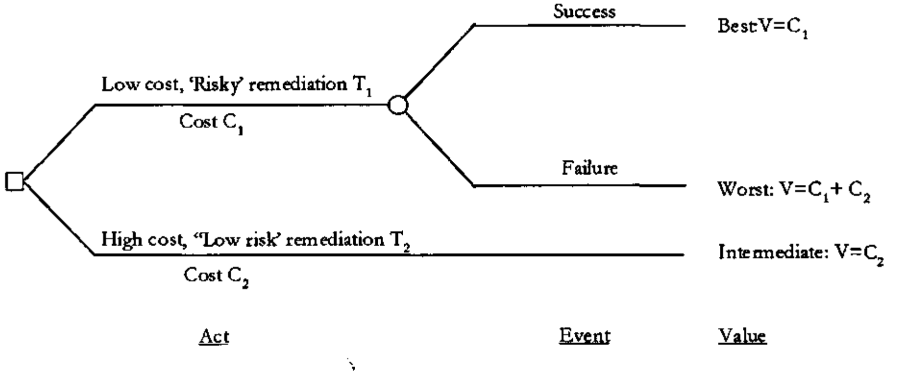

Figure 3.3. A decision tree for taking over a possibly contaminated site

An immediate choice for T_2 results in intermediate costs. Costs are known for each branch, but a final decision can be reached only if these costs are combined with probabilities of events. The best decision is a choice for the technique with the highest value.

As a second example, decisions for clean-up technologies rely for a large part on prior information such as surveys, laboratory tests or pilot tests. The choice for relevant prior information is made before deciding upon the clean-up technology, yielding a sequence of decisions. Figure 3.4 shows how we may first run a test with costs C_3 before deciding upon either T_2 or T_1. Costs can be calculated for each branch, but the final decision can only be made after combining economic considerations with event probabilities.

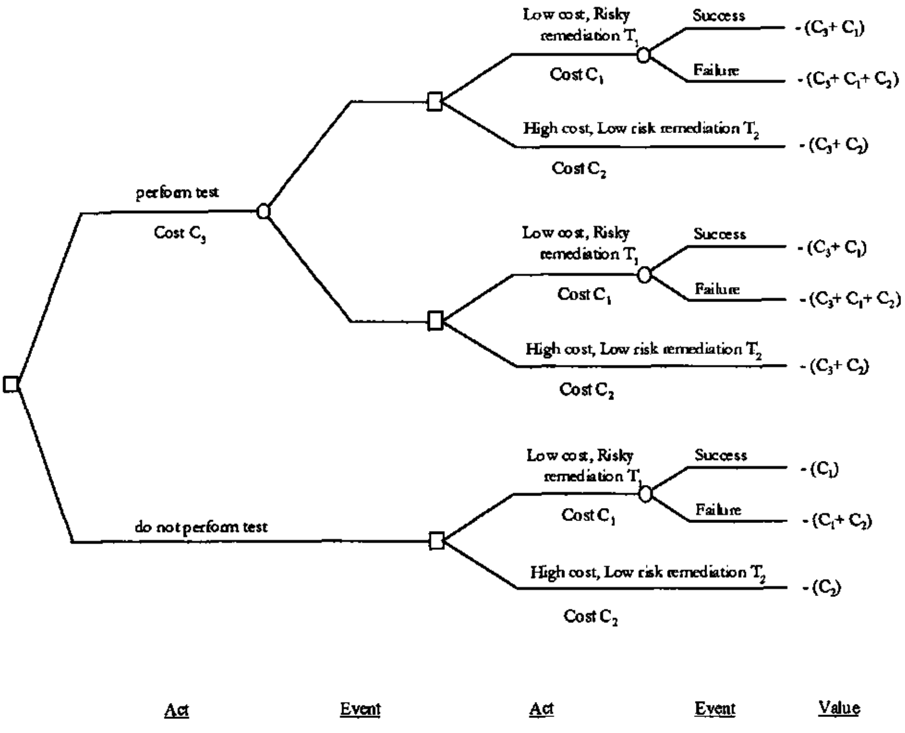

Figure 3.4. A decision tree for several options on clean-up technologies

3.4 Results

The take-over case

In the take-over case the company has to choose between two survey schemes. Missing circular (S=1) hot spots with a radius of 12.5 m will result in unforeseen remediation costs of approximately 90 kECU. Strategy T_1 is given by a scheme designed for total certainty, i.e. *Pr (Hot spot is detected) =1*, e.g. of a consumer's risk β equal to 0, whereas strategy T_2 is equal to a scheme that accepts a consumer's risk equal to 0.1. The necessary grid spacings are 17.6 m for T_1 and 22.3 m for T_2, requiring 194 and 122 samples with associated costs of 17.6 kECU and 11.1 kECU, respectively. Therefore the company either pays for the costlier, but never failing T_1 or cuts costs and decides for T_2 (Figure 3.5). The value of T_1 equals $1 \times 17.6 + 0 \times 107.6 = 17.6$ kECU, whereas the value of T_2 equals $0.9 \times 11.1 + 0.1 \times 101.1 = 20.1$ kECU. Therefore, the value of T_1 is lower than that of T_2, although an initial glance at the sampling costs may have suggested differently.

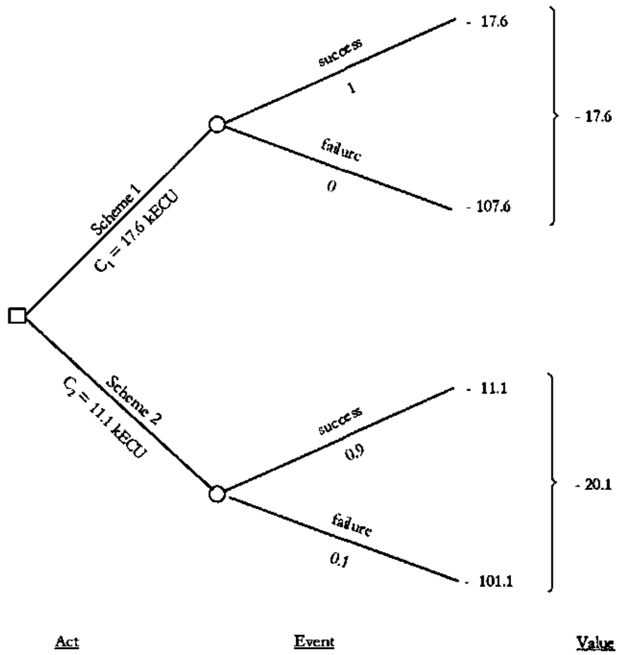

Figure 3.5. The decision tree for the take-over case. Costs and value in kECU

Table 3.1. Results of the sample survey (n = 20)

	Lead (mg.kg^{-1})	Clay content (%)	Organic matter content (%)
Mean	305	5.3	4.0
Variance	60516	7.3	13.7
Standard error	55.0	0.6	0.8

The metal pollution case

For the metal pollution case, the decision tree is given in Figure 3.6, whereas summary statistics for the lead observations are given in Table 3.1. Use of mean clay and organic carbon contents yields $I_{Pb} = 6.24 \times (50 + 5.3 + 4.0) = 370$ mg·kg^{-1} as the intervention level. We want to know whether the observed sample mean m of 305 mg·kg^{-1} Pb indicates a soil quality below the intervention value.

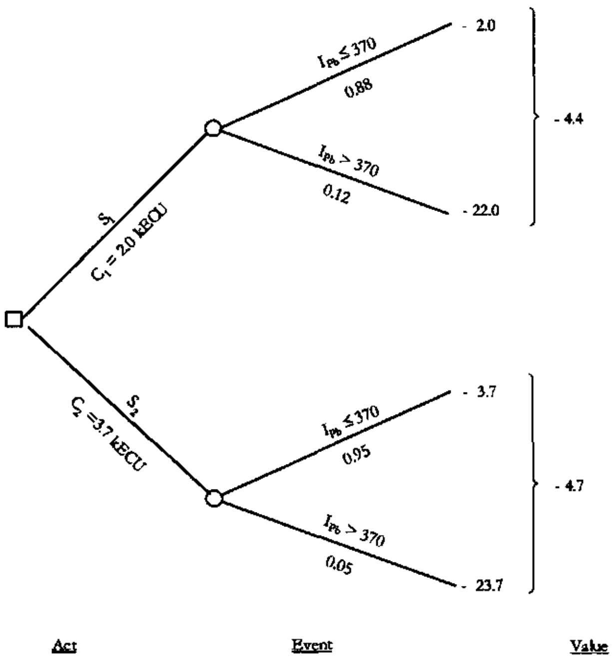

Figure 3.6. The decision tree for the metal pollution case. Costs and values in kECU

The probability of exceeding the intervention value can equals

$$\Pr(m > 370) = \Pr\left(\frac{m-305}{55} > \frac{65}{55}\right) = 0.119,$$

assuming that m follows a standard Gaussian distribution. The number of samples needed to reduce the probability of exceeding I_{Pb} to 0.05 equals 37. Recall that exceeding the intervention value would require an additional investment of 20 kECU and an additional sample requires 0.1 kECU. The decision maker therefore has to decide whether take the risk for granted (T_1) or to invest into an additional 1.7 kECU sample survey (T_2). The values of the two decisions are $0.88 \times 2.0 + 0.12 \times 22.0 = 4.4$ kECU for T_1 and $0.95 \times 3.7 + 0.05 \times 23.7 = 4.7$ kECU for T_2. Thus, in this case, the 20 sample survey is the better choice and there is no need for a further investment.

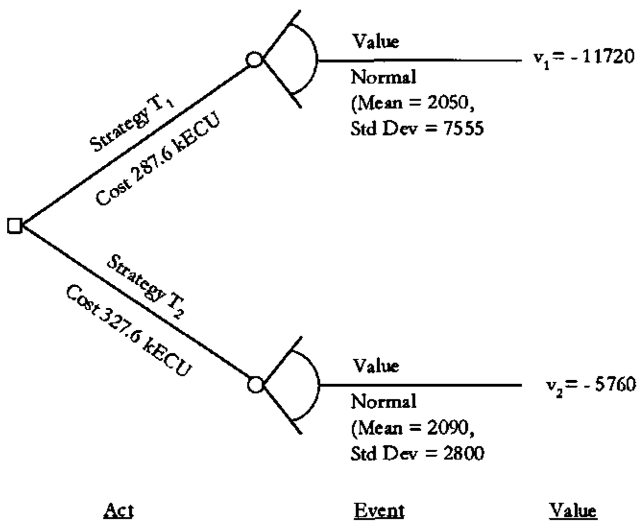

Figure 3.7. The decision tree for the case of taking additional samples to delineate a volume of contaminated soil. Costs and values in kECU

The case of taking additional samples to delineate a volume of contaminated soil

The average outcome using probability kriging on 719 samples (T_1) shows a volume above the threshold equal to 2050 m^3. As the clean-up of 1 m^3 costs approximately 1 kECU, expected costs (E_i) equal 2050 kECU. Taking the normal distribution N[2050,7555] of the amount of volume to be removed, and using a k-factor of 1.28 allowing 10% chance of exceeding v_i, the costs are set to 11720 kECU. To reduce the uncertainties additional sampling of 100 samples (T_2) at a cost of 40 kECU is evaluated. The additional samples are not to be equally distributed over the site, but are to be taken at a small distance of the locations were the 5 $mg.kg^{-1}$ Cd boundary is predicted with the original 719 observations and were the kriging standard-deviation is highest. These are on the one hand the locations of the highest interest with values close to t_z, and on the other hand with the largest uncertainty. With the additional sampling the normal distribution changes to N[2090,2800], and using the cost formula with k-factor 1.28, the costs calculations yield 5760 kECU. It has to be decided whether the costs calculated with the original strategy (T_1) are acceptable or whether additional sampling (T_2) should be preferred. The decision tree is given in Figure 3.7. Even though the expected costs attached to T_1 are lower than the expected costs attached to T_2, T_1 has a far greater probability of costs higher than 5600 kECU. To avoid excessive costs the additional sampling (T_2) should be carried out.

The case of choosing the best clean-up treatment

We calculate the conditional probability that *in situ* treatment is possible for the 4th case given a positive column test result. Recall that the expected costs for *ex situ* remediation equals 500 kECU. We will now investigate possible options for *in situ* remediation. Define the following decision alternatives:

T_1 = No a priori tests;

T_2 = Use of a column test;

T_3 = Use of a pilot plant.

The event A is the event that clean soil emerges after *in situ* remediation, the event B is a positive indication given by either the column test or the pilot plant for *in situ* remediation. Figure 3.8 gives an overview of possible outcomes of the project with a choice between column test and pilot plant as an option as well. Starting with the payoffs listed on the right-hand side of the diagram we can reduce event forks to one single expected cost.

For T_1, Pr(A) = 0.5 and its value V_1 expressed as expected costs equals $0.5 \cdot 250 + 0.5 \cdot 750 = 500$ kECU.

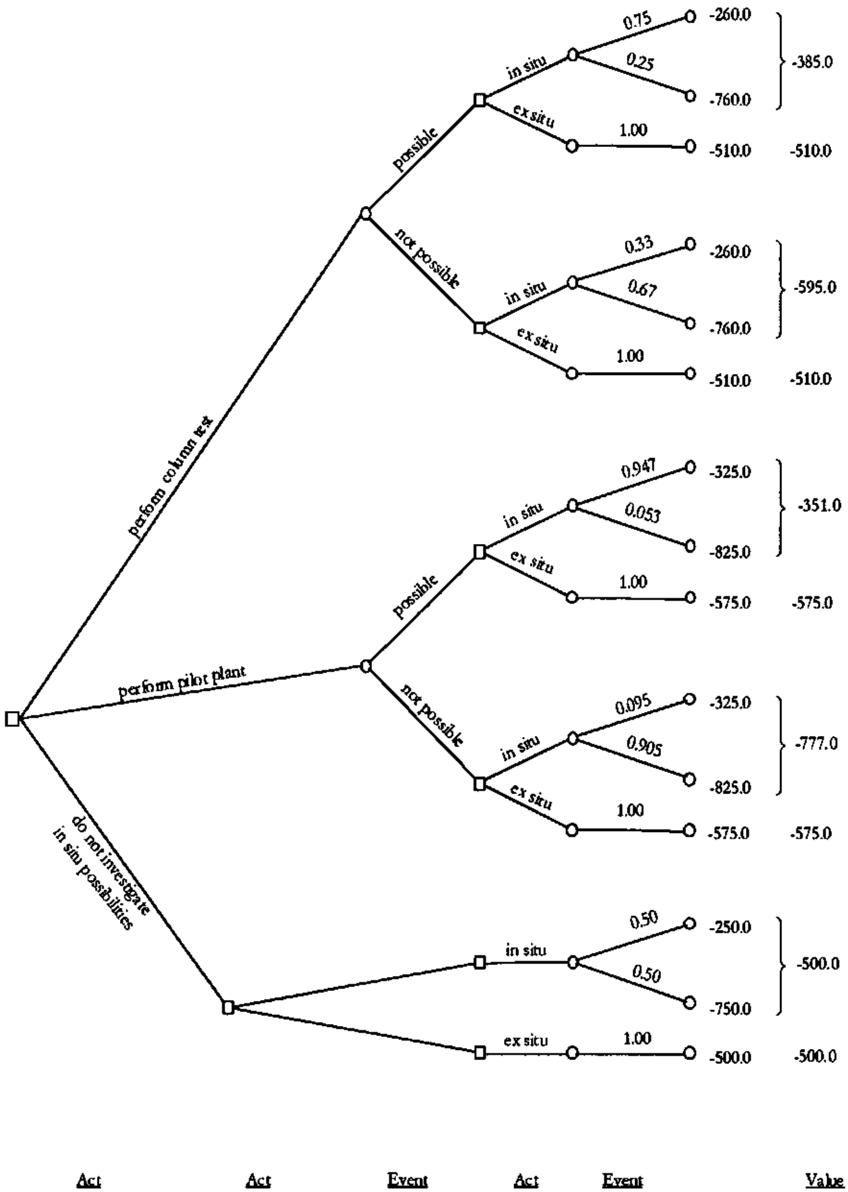

Figure 3.8. The decision tree for choosing the best clean-up treatment. Costs and values in kECU

For T_2, $Pr(A\,|\,B)$= 0.6, $Pr(A\,|\,\overline{B})$=0.2, $Pr(\overline{A}\,|\,B)$=0.4 and $Pr(\overline{A}\,|\,\overline{B})$ = 0.8. We therefore have that $Pr(AB)$= 0.3, $Pr(A\overline{B})$= 0.1, $Pr(\overline{A}B)$= 0.2 and $Pr(\overline{A}\overline{B})$= 0.4. The probability that clean soil emerges and a positive column test equals 0.3/0.4 = 0.75. Thus the value V_2 of T_2 expressed as expected costs equals 10 + 0.3·250 + 0.2·(250 + 500) = 372.5 kECU. We notice, therefore, that the value of T_2 increases as compared to T_1.

A similar reasoning applies for T_3. The probability of obtaining clean soil and a positive pilot plant equals 0.45/0.475 = 0.947. Therefore use of the pilot plant yields a higher value for T_3, and lower expected costs, being equal to 75 + 0.947·250 + 0.053·(250+500) = 351.3 kECU. These calculations show that an investment in a column test is useful and beneficial, whereas investment in a pilot plant is even more so.

3.5 Discussion

In this chapter use has been made of statistical decision trees to value investigation strategies. These trees have been presented for two levels of random events. Without any problem they can be extended to more levels as well. The decision trees have the main advantage of allowing to use available prior information and of existing data in an optimal way. The decisions, however, largely rely upon sound quantitative knowledge of conditional and unconditional probabilities for occurrence of events that may influence the decision. A relevant question may be whether these probabilities can reliably be estimated from previous experience. Moreover, it is as yet unclear how uncertain guesses for these will influence the decisions to be made.

In this chapter the value of a strategy is expressed in kECU, whereas, in the literature it is often referred to as a measurement of relative *liking* or *preference* on the part of a decision maker for particular outcomes. We have assumed a linear relation between kECU and value. This is not always a realistic assumption. Decision makers can have three different attitudes towards risk: risk avoiding, neutral or risk seeking. These attitudes can be expressed as preferences or as utility functions. Determination of these functions is described in most textbooks on the subject. Once we have obtained the preference or utility function we can transform the ECU's in preferences or liking.

Another issue concerns the degree to which decision trees can be introduced to value decision strategies. Although they give a relatively clear picture of events, actions and probabilities, there is still a large uncertainty on using correct and precise estimates of conditional and unconditional probabilities. On the one hand they may be obtained from case databases that are widely available at engineering offices. On the other hand, the value of the strategies may be relatively insensitive to minor changes in probability values.It is our experience so far that decision trees serve well to identify uncertainties in a decision strategy, and to quantify their values, whereas an indication on the accuracy of the outcomes is usually appreciated.

3.6 Conclusions

Statistical decision trees are useful to value investigation strategies for soil remediation problems. They require to specify probabilities and to properly distinguish between decisions and random events. They allow to calculate expected payoffs or costs of a remediation strategy. Decision trees give insight into possible decision strategies and are simple to apply. Decision trees allow a careful reflection on specific probability and value inputs. To evaluate conditional decisions we can make use of probabilistic reasoning. They do not provide the answer directly, however, but present an overview of consequences of choices for any realistic option. In this chapter decisions for four common problems have been supported successfully by using decision trees: whether a hot spot is present, how much of a contaminant is present, where is the contaminant present and is it useful to invest in a pilot study. We have shown that (cheap) strategies do not always lead to the lowest costs or highest payoff. Valuing sampling schemes or experimental set-ups proves to be useful.

Appendix

To find a hot spot one has to decide for the grid spacing needed to hit it with specified confidence. Singer (1975) gives solutions for locating geologic deposits by sampling on a square, rectangular, or triangular grid. The hot spot must be characterised by an ellipse with the length of its longest axis equal to l_1 and of its shortest axis equal to l_2. Its shape equals $s = \frac{1}{4} \cdot \pi \cdot l_1 \cdot l_2$. The probability ($P_{detect}$) of detecting a hot spot is equal to the area of the hot spot divided by the area of the sampled unit. If the hot spot is similar to a circle with diameter L, and we apply a square grid with grid spacing G we obtain a simple

$$P_{detect} = \frac{\text{area of pollution}}{\text{area of unit}}$$

thus

$$P_{detect} = \frac{\pi L^2}{G^2} - \frac{2L^2}{G^2}(2\theta - \sin(2\theta))$$

where

$$\theta = \arcsin\left(\sqrt{1 - \frac{G^2}{4L^2}}\right).$$

If $L/G \geq 1/\sqrt{2}$, then $P_{detect} = 1$. With this equation it is now easy to relate L/G to the consumer's risk β (Figure 1). The analytical solutions for rectangular and triangular grids are similar. Note that the probability of not detecting a hot spot, commonly addressed as the consumer's risk ß, is equal to 1 - P.

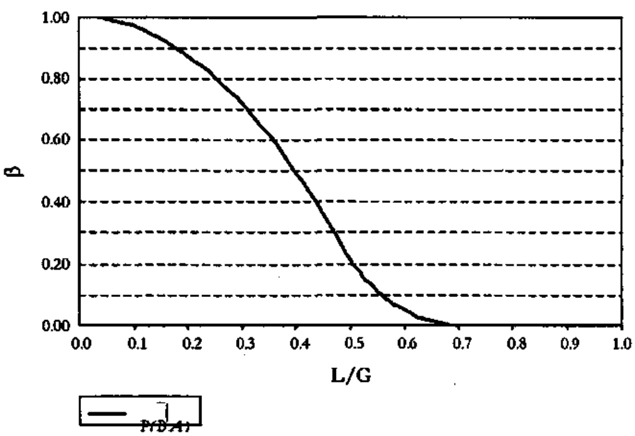

Figure 1. The consumer's risk β as a function of L/G

Chapter 4

PROBABILITY KRIGING FOR LOCAL SOIL REMEDIATION PROBLEMS

In a soil remediation project, the amount of polluted soil that will be removed seldom equals the total amount of polluted soil. Nevertheless an estimate of the amount of polluted soil is essential for the development of remedial scenarios. It is the key to cost calculations.

Probability kriging is a non-linear geostatistical estimation technique suitable for the estimation of the amount of polluted soil material. It provides conditional probabilities of exceeding user-defined thresholds. For volume estimation purposes, two pieces of information are essential. First, concentration levels of the contaminants related to different remedial actions or to different policy-views have to be known. These thresholds are called cut-off levels. Second, the probability level above which a remedial action becomes inevitable has to be chosen. The possibility of working with more than one cut-off level makes the technique an important management tool in decision-making.

In the described case studies probability kriging was used to map the heavy metal contamination in terms of conditional probabilities that given thresholds are exceeded.

In the first case study it was found that in a large area of about 30 km^2 around a pigment factory the probability of exceeding the 150 $mg.kg^{-1}$ threshold for lead is over 0.8. The pattern of concentric circles around the factory obtained when estimating the probabilities of exceeding the 300 $mg.kg^{-1}$ threshold for lead is what is expected in a case of atmospheric deposition when taking the wind direction in account.

In the second case study it was found that approximately 5000 m^3 soil exceeds the 2.5 $mg.kg^{-1}$ threshold with probability equal to 0.5 or higher and approximately 2000 m^3 soil exceeds the 5.0 $mg.kg^{-1}$ threshold for cadmium with the same probability.

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Wahrscheinlichkeitskrigen für Bodensanierungszwecke. Okx, J.P., Kuipers, B.R. and Süßkraut, G.. In: *Beiträge zur Mathematischen Geologie und Geoinformatik. Band 5. Neue Modellierungsmethoden in Geologie und Umweltinformatik* Peschel, G.J. (ed.), Verlag Sven von Loga, Köln.

Dat het onbekende als zodanig, zonder verdere qualificatie, angst kan opwekken, heb ik reeds eerder bestreden. Het onbekende is in dit opzicht volstrekt neutraal; het kan gevaarlijk zijn, het kan het ook niet zijn, en zolang het onbekend blijft, beschikken wij niet over de middelen om uit deze twee mogelijkheden een keuze te doen. Maar waarschijnlijk stoort de menselijke natuur zich des te minder aan deze logische onaantastbare uitspraak, naar gelang de ervaring leert, dat het onbekende vaak wel degelijk gevaarlijk is, en dan ook in verdubbelde mate, omdat het tevens het onbeheersbare is. [4]

Probability kriging as a decision support tool for local soil pollution problems

4.1 Introduction

In a soil remediation project the total volume of polluted soil that has to be removed is the key factor in the cost calculations. These volumes are estimated from the isarithmic maps on which the results of the soil pollution research are presented. However, in practice, the estimated volumes are often exceeded. Normally this is not caused by poor estimation but by not taking into account the errors associated with the estimates. In many cases these errors are not known; simply because the non-statistical interpolation routines used do not provide them. Fortunately there are now a number of geostatistical techniques available which are not only better estimators than the methods normally used in pollution studies but they also give the errors related to the estimates. The work of Krige (1951) and Matheron (1965) should be considered as the basis for this work.
So the main advantage of geostatistical interpolation techniques, essentially ordinary kriging, is that an estimation variance is attached to each estimate. Unfortunately, unless a Gaussian distribution of spatial errors is called for, an estimation variance falls short of providing confidence intervals and the error probability distribution required for risk assessment. Regarding the characterisation of uncertainty, most interpolation algorithms , including kriging, are parametric in the sense that a model for the distribution of errors is assumed. Such models are questionable when used for spatial interpolation errors (Journel, 1987).

[4] Taken from Simon Vestdijk, "Het wezen van de angst" (1949), Uitgeverij De Bezige Bij, Amsterdam

Non-parametric geostatistical methods, such as indicator kriging (Journel, 1983) and probability kriging (Sullivan, 1984 and Journel, 1984), which puts the modelling of the uncertainty as priority, do not make use of these assumptions.
Since probability kriging is considered as an improvement of the indicator kriging procedure in the sense that the data is used more completely, we decided to use probability kriging.
One of the ways to present the results of the probability kriging procedures is by means of a probability map on which conditional probabilities of exceeding given thresholds are represented. This way of representation makes the technique an important tool in decision making under risk because it communicates the uncertainties to decision-makers such as the responsible authorities or problem-owners (Okx and Kuipers, 1991). After a short description of the method the advantages will be demonstrated in the form of a case study.

4.2 Probability kriging

Probability kriging is a non-parametric procedure that does not depend on any particular statistical distribution. It is based on an indicator function $I(x_\alpha, z)$, which is then used for estimating a local distribution . The indicator function is defined in terms of the cut-off value z that is a threshold given by environmental protection regulations. If the values of the property are below the threshold no action is required, but if then certain measures may become necessary.

The indicator function is:

$$I(x_\alpha, z) = \begin{cases} 1 \text{ if } z(x_\alpha) > z \\ 0 \text{ else} \end{cases}$$

In this article a short description of probability kriging is given and we recommend for further information Sullivan (1984) and Journel (1984). As stated earlier, probability kriging is an improvement on indicator kriging. Indicator kriging (see Journel, 1983) does not use all the available information about the variable to be estimated. When the probability of exceeding a certain cut-off value, $(P(z))$, is estimated only the indicator values of that particular cut-off value (z) are used.
The basic idea is to use the grade information $Z(x_\alpha)$ in addition to the indicator $I(x_\alpha, z)$. The values of $Z(x_\alpha)$ and $I(x_\alpha, z)$ are usually of a different magnitude, which may give numerical problems. To overcome this problem the grades are replaced by their cumulative distribution function. We will use a variable $U(x)$:

$$U(x_\alpha) = F(Z(x_\alpha))$$

Now the estimated value is:

$$\varphi^*(V,X) = \sum_{i=1}^{n} \lambda_\alpha \; I(x_\alpha, z) + \sum_{i=1}^{n} \nu_\alpha \; U(x_\alpha)$$

Weights λ_α and ν_α will be obtained from a standard cokriging system:

$$\sum_{i=1}^{n} \lambda_\alpha \; \rho_i(x_\alpha - x_\beta, Z) + \sum_{i=1}^{n} \nu_\alpha \rho_{ui}(x_\alpha - x_\beta, Z) + \mu_1 = \rho_i(x_\beta, V, Z)$$

$$\sum_{i=1}^{n} \lambda_\alpha \; \rho_{ui}(x_\alpha - x_\beta, Z) + \sum_{i=1}^{n} \nu_\alpha \rho_u(x_\alpha - x_\beta, Z) + \mu_2 = \rho_{ui}(x_\beta, V, Z)$$

$$\sum_{i=1}^{n} \lambda_\alpha = 1$$

$$\sum_{i=1}^{n} \nu_\alpha = 0$$

(David, 1988).

4.3 Case study: athmospheric deposition of lead

Source of contamination and sampling scheme

The soil in areas surrounding the town of Eijsden in the Province of Limburg in The Netherlands has been polluted with heavy metals in several ways. Apart from the diffuse inputs through atmospheric deposition and the use of fertilisers containing metals, three site specific sources of heavy metals can be distinguished (Rang et al., 1987).

One source is a pigment-producing factory, which was established in Eijsden in 1870. At the beginning of this century the factory emitted large quantities of particles containing barium, zinc and lead into the atmosphere. In 1974 the annual rate of emission was assessed to be 64 tons of zinc, 210 tons of barium and 94 tons of lead. In 1975 the emission stopped almost completely as a result of changes in the production process.

A second site specific source of heavy metal pollution is from roads sealed with zinc oxide cinders, which are a waste product of the pigment industry. Between 1870 and 1975, 250,000 tons of cinders were produced. The total length of cinder-sealed roads in the area is about 100 km.
A third source of heavy metals is the river Meuse. Part of the area of investigation is located on the floodplain of the river. The river mud's are contaminated with heavy metals as a result of waste disposal by mines and metallurgic industries in the Belgian part of the catchment. Flooding and subsequent deposition of these mud's has resulted in a considerable increase in the heavy metal concentrations of the topsoil in the floodplain area.

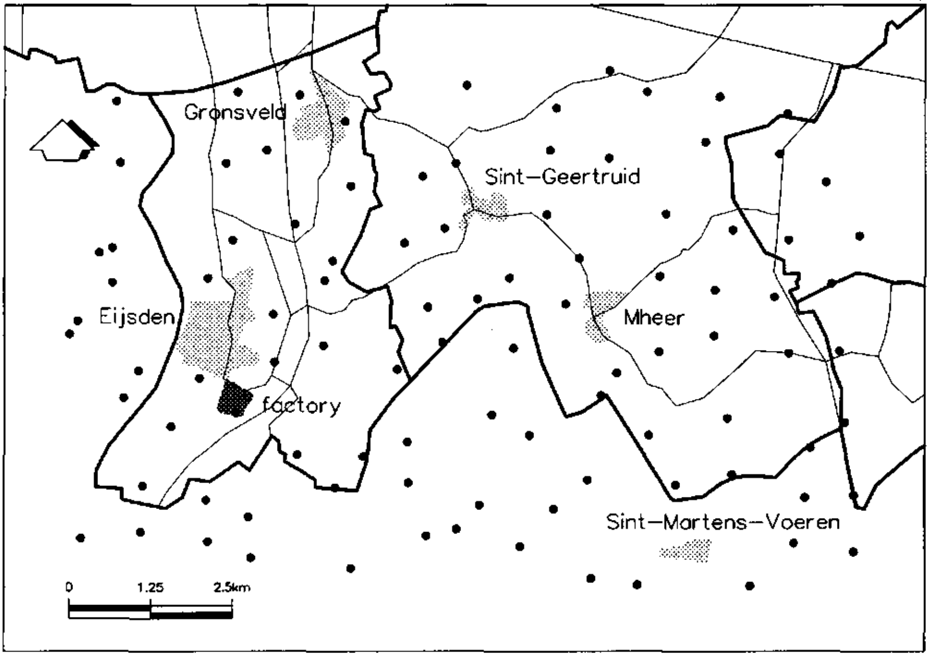

Figure 4.1. The sampling scheme for the pigment factory

To estimate the spatial extent of soil pollution by atmospheric emissions from the pigment factory, 98 samples were taken from the topsoil in the area. To avoid disturbing the pattern of atmospheric deposition, random stratified sampling restricted to areas where the presence of cinders or river-sediments was unlikely was used. The strata were obtained by dividing the area into 1 x 1 km^2 squares. In this case study we focused on the lead content of the soil because lead turns out to be the most critical element in terms of human-toxicological and ecotoxicological risk. The sampling scheme is given in Figure 4.1.

Summary statistics

The statistical analysis, which should be the starting point of any geostatistical study, gives us a first impression of the variability of the variable in study. Table 4.1 shows the most important summary statistics.

Table 4.1. Summary statistics of the lead content

Number of samples	Min ($mg.kg^{-1}$)	Max ($mg.kg^{-1}$)	Mean ($mg.kg^{-1}$)	Median ($mg.kg^{-1}$)	standard-deviation	Coefficient of variation	Skewness
98	83	5640	248	155	570	2.30	9.0

The large value of 2.30 for the coefficient of variation illustrates the considerable variability in the lead content of the soils in the area. A maximum value of 5640 $mg.kg^{-1}$ is reached. This means that the soil is extremely polluted locally which calls for remedial action. The minimum value of 83 $mg.kg^{-1}$ shows that all values are well above the detection limit of 10 $mg.kg^{-1}$ so there is no zero effect. The frequency histogram (Figure 4.2) is markedly asymmetric and this is also reflected by the large skewness-value of 9.0. This distribution clearly does not follow any gaussian or lognormal law, a fact that is not unusual for soil pollution data.

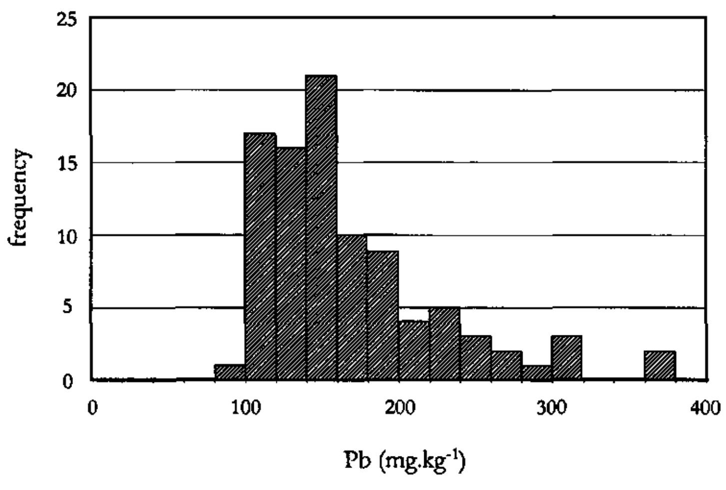

Figure 4.2. Frequency histogram of lead content

The first step in the probability kriging procedure is to define a number of cut-offs. Different cut-offs could be related to a number of different policy-views. These views can be based on:

- risk assessment
- technical possibilities
- economical context
- political context.

In Table 4.2 two different cut-offs (z_a) and their corresponding transformations $(U(z_a))$ are given. These $U(z_a)$ values correspond to the relative cumulative frequency of the distribution function $H(z(x_i))$.

Table 4.2. Cut-offs z_a and the corresponding transformations $(U(z_a))$

cut-off (mg.kg^{-1}) Z_a	150	300
$U(Z_a)$	0.469	0.915

Table 4.3. Parameters of the spherical models of the different variograms

variogram	C_o	C_1, a_1	C
I_{150}	0.150	0.293, 7500	0.51
I_{300}	0.020	0.093, 3500	0.21
U	0.015	0.091, 7500	0.16
UI_{150}	0.045	0.150, 7800	0.30
UI_{300}	0.001	0.039, 3500	0.03

The model parameters (c_0: nugget effect: c_1, a_1: sill and range: c'_0: relative nugget effect) are given in Table 4.3. The table shows the model parameters of the two indicator variograms (I_{150},I_{300}), of the variogram of the transformed sample values, U, and of the two cross variograms, UI_{150} and UI_{300}. The experimental variograms I_{150}, U and UI_{300} all start with relatively high values, but they are based on two sample pairs only. Since the parameters of the variogram models are fitted by a simple weighted least squares approximation (Cressie, 1985) these starting points are practically neglected.

In the process of cross variogram fitting the Cauchy-Schwartz inequality:

$$\left|\left(\gamma_{ui}(h)\right)\right| \leq \sqrt{\left(\gamma_{u}(h) * \gamma_{i}(h)\right)} \textit{ for all } h \geq 0$$

was checked to guarantee the positive-definiteness of the kriging matrices (Journel and Huybregts, 1978; Myers, 1982 and 1984; Nienhuis, 1987). It satisfies the inequality because the square of the covariance of increments from two variables is bounded by the product of the corresponding increment variances (Wackernagel, 1995).

The cross variogram models (UI_{150} and UI_{300} were used for probability kriging which resulted in a number different blocks (125 x 125 m^2) filled with the probability that the cut-offs in question (150 $mg.kg^{-1}$ and 300 $mg.kg^{-1}$) is being exceeded. Figures 4.3 and 4.4 show the results for the topsoil in the Eijsden-area, the probabilities are represented as percentages.

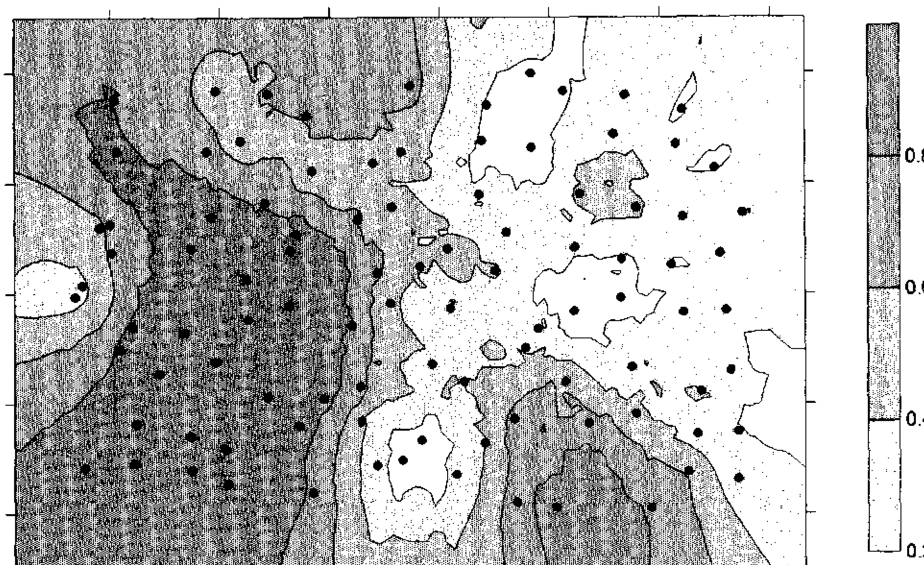

Figure 4.3. Conditional probabilities of exceedence of the 150 $mg.kg^{-1}$ threshold for lead

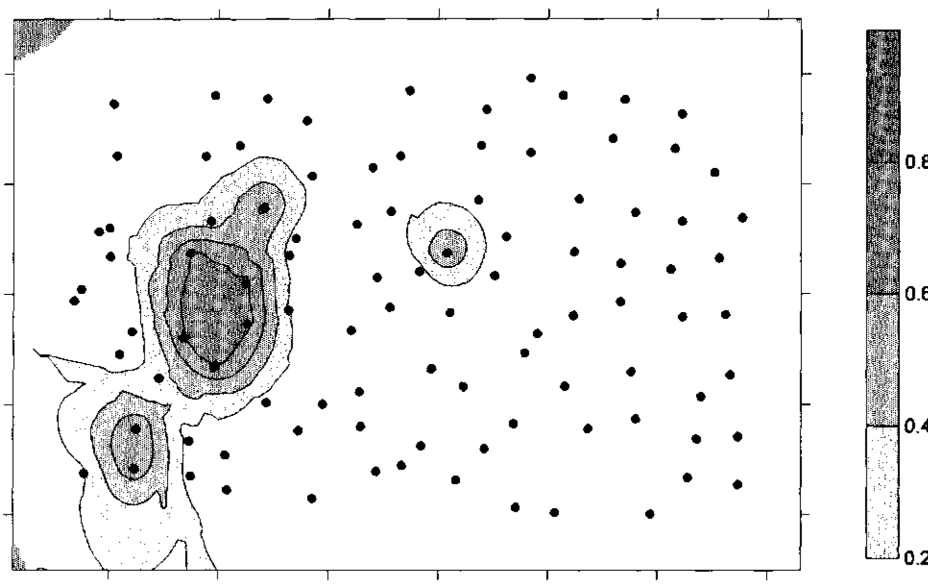

Figure 4.4. Conditional probabilities of exceedence of the 300 mg.kg^{-1} threshold for lead

Figure 4.3 of the probabilities of exceeding the 150 mg.kg^{-1} threshold shows that there are two distinct areas. One large area around the pigment factory of about 30 km^2 with probabilities of above 0.8 that the 150 mg.kg^{-1} threshold is exceeded that has probably been exposed to atmospheric deposition from the factory. In the rest of the area the probabilities of exceeding the 150 mg.kg^{-1} threshold are highly variable and range from 0.1 to 0.7.

In Figure 4.4 the probabilities of exceeding the 300 mg.kg^{-1} threshold are mapped and the relation with our factory is even clearer. The pattern of concentric circles around the factory is what you would expect in a case of atmospheric deposition. The area where the probability of exceeding the 300 mg.kg^{-1} threshold is over 0.8 is approximately 2.4 km^2.

By using simple tools as provided by geographical information systems we can easily calculate the consequences - in terms of the area for which certain measures could become necessary - for each possible combination of a cut-off and a critical probability level. This makes the technique an important management tool in the decision making process. The results of such calculations are given in Figure 4.5. The

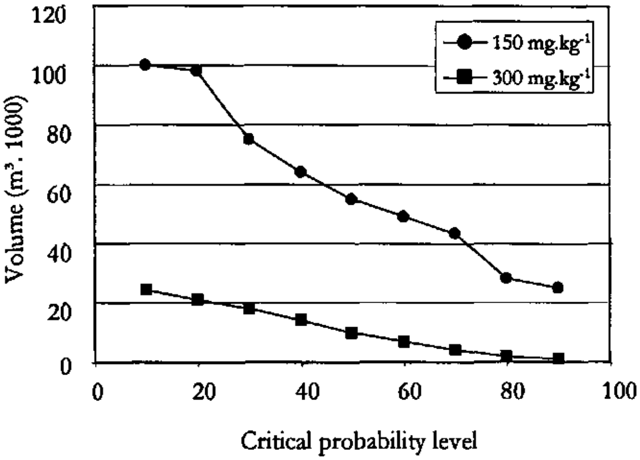

Figure 4.5. Area in which measures could be necessary as a function of cut-off and a critical probability level

upper line shows the relation for the 150 mg.kg^{-1} threshold and the lower line for the 300 mg.kg^{-1} threshold. If we consider the probability levels in Figure 4.5 as being consumer's risks, then it becomes clear that normally remedial action is a matter of reducing the consumer's risk rather than eliminating it.

4.3 Discussion

Probability kriging is an essential tool in the decision making process. The ability to produce probability maps greatly enhances the possibility of communicating uncertainties to the authorities responsible for soil clean-up operations. At present the decision-makers such as responsible authorities or problem-owners are not used to think in terms of probabilities and risks and it is therefore unlikely that critical probability levels will be specified a priori. The concept of a remedial action that reduces risk instead of eliminating it will force the decision-makers to make formal statements about what should be considered as an acceptable risk. Just as choosing different cut-offs can be based on different aspects such as risk assessment, and economic and political contexts, making statements about acceptable risks is also a matter of taking into account and weighing different aspects.

Information on the local background levels - preferably also in the form of conditional probabilities - is, in our opinion, essential in formulating statements that lead to decisions in which costs and efficiency are well balanced.

The non-parametric nature of this method is yet another advantage when analysing highly skewed phenomena that are so frequently encountered in soil pollution studies of this kind.

Nicht Kunst und Wissenschaft allein,
Geduld will bei dem Werke sein.
Ein stiller Geist ist Jahrelang geschäftig,
Die Zeit nur macht die feine Gärung kräftig ...[5]

Wahrscheinlichkeitskrigen für Bodensanierungszwecke

4.5 Einleitung

Je mehr Bodenmaterial ausgekoffert werden muß, desto höher fallen die Sanierungskosten aus. Die für die Einschätzung des Volumens notwendigen Zahlen gehen aus den Karten hervor, in denen die Resultate der Verunreinigungsuntersuchung aufgezeichnet sind. Nun kommt es häufig vor, daß sich das auszukoffernde Volumen in der Praxis als größer erweist, als theoretisch eingeschätzt worden war.

Dies bedeutet in der Regel aber nicht, daß die Schätzungen an sich von ungenügender Qualität waren, sondern vielmehr, daß den mit der Schätzung zusammenhängenden Unsicherheiten nicht Rechnung getragen wurde. Dies kommt vor aus dem einfachen Grund, daß die übliche Interpolationsverfahren diese Unsicherheiten nicht auswiesen. Mittlerweile haben Geostatistiker aber eine Reihe von Techniken entwickelt, die nicht nur bessere Schätzmethoden, sondern auch Aussagen über die mit der Schätzung verbundenen Unsicherheiten liefern. Als Grundlage für diese Entwicklungen dienten die Arbeiten von Krige (1951) und Matheron (1965).

Der wichtigste Vorteil der geostatistischen Interpolationsverfahren, wie unter anderen da sogenannte normale Krigen, ist daß eine Schätzungsvarianz mit jeder Schätzung verbunden wird. Unglücklicherweise sind diese Schätzungsvarianzen nicht geeignet um irgendwelche Vertrauensintervalle oder Wahrscheinlichkeitsverteilungen zu ermitteln oder um Risikoeinschätzungen zu errechnen. Gewöhnlich setzt man die Gauß'sche Verteilung der räumlichen Schätzungsfehler voraus. Anläßlich der Charakterisierung von Unsicherkeiten sind die meisten Interpolationsmethoden, Krigen einschließlich, parametrischer Art. Das heißt, daß ein gewisses Modell für die Verteilung von Schätzfehler angenommen wird. Diese Modelle sind aber alle fraglich, wenn sie für das Einschätzen von Interpolationsfehlern verwendet werden (Journel, 1987).

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[5] Johann Wolfgang von Goethe, "Faust" (1808)

Nicht-parametrische geostatistische Methoden, so wie Indikatorkrigen (Journel, 1983) und Wahrscheinlichkeitskrigen (Sullivan, 1984; Journel, 1984), die beiden das Modellieren der Unsicherkeiten als erste Priorität haben, gehen nicht von irgendeiner Verteilung aus. Weil Wahrscheinlichkeitskrigen als eine Verbesserung des Indikatorkrigens betrachtet wird, weil es die vorhandenen Daten besser ausnutzt (David, 1988), werden wir in diese Verhandlung das Wahrscheinlichkeitskrigen verwenden.

Eine der Möglichkeiten, die Resultate des Wahrscheinlichkeitskrigens zu präsentieren, ist die Wiedergabe einer Wahrscheinlichkeitskarte; sie zeigt bedingte Wahrscheinlichkeiten auf, mit denen im voraus festgelegte Schwellenwerte überschritten werden. Diese Art der Präsentation macht diese Technik zu einem nützlichen Instrument, wenn es in Risikosituationen Entscheidungen zu fällen gilt, da sie dem Entscheidungsträger auch die Unsicherheiten verdeutlicht. Der Vorteil dieser Methode wird in Form einer Einzelfallstudie aufgezeigt.

4.6 Wahrscheinlichkeitskrigen

Das Wahrscheinlichkeitskrigen ist ein nicht-lineares Verfahren, das nicht an eine bestimmte Verteilung gebunden ist. Es basiert auf der Definition einer sogenannten Indikatorfunktion $I(x_i;z_c)$, die für die Einschätzung einer lokalen Verteilung verwendet wird.

Der sogenannte Cutoff-Wert z_c ist ein vom Anwender festgelegter Schwellenwert. Unterhalb dieses Wertes sind keine Maßnahmen erforderlich, übersteigen die festgestellten Werte diese Schwelle, dann sind Maßnahmen erforderlich.

Die Indikatorfunktion ist dann:

$$I(x_\alpha, z) = \begin{cases} 1 \; wenn \; z(x_\alpha) > z \\ 0 \; sonst \end{cases}$$

Der folgende Abschnitt beschreibt das Vorgehen des Wahrscheinlichkeitskrigens. Zur weiteren Lektüre werden Sullivan (1984) und Journel (1984) empfohlen. Das Wahrscheinlichkeitskrigen kann als Verbesserung des Indikatorkrigens betrachtet werden. Das Indikatorkrigen wurde von Journel (1983) beschrieben; es macht nicht von allen verfügbaren Informationen über die einzuschätzende Variable Gebrauch.

Die Indikatorwerte anderer Cutoff-Werte korrelieren auch mit den Werten, die geschätzt werden müssen, so daß auch sie im Rahmen der Einschätzung nützlich sein können. Zu diesem Zweck wäre die Anwendung des klassischen Koindikatorkrigens mit Hilfe eines Cross-Indiktorvariogrammes nützlich; allerdings wäre dieses

Vorgehen eher schwerfällig. Aus diesem Grund werden beim Wahrscheinlichkeitskrigen nebst den Indikatorwerten auch die tatsächlichen Proben-Werte benützt, um die Funktion der kumulativen Dichte der Konzentrationen in Bezug auf die Cutoff-Werte einzuschätzen. Die Werte $Z(x_\alpha)$ und $I(x_\alpha,z)$ sind aber normalerweise nicht in derselben Großenordnung, deshalb, wird eine Umwandlung der Probenwerte

$$U(x_\alpha) = F(z(x_\alpha))$$

Der Schätzungswert folgt dann aus:

$$\varphi^*(V,X) = \sum_{i=1}^{n} \lambda_\alpha \; I(x_\alpha, z) + \sum_{i=1}^{n} \nu_\alpha \; U(x_\alpha)$$

Die Wichtungen λ_α und ν_α folgen aus einem Standard-Kokrige-System:

$$\sum_{i=1}^{n} \lambda_\alpha \; \rho_i(x_\alpha - x_\beta, Z) + \sum_{i=1}^{n} \nu_\alpha \rho_{ui}(x_\alpha - x_\beta, Z) + \mu_1 = \rho_i(x_\beta, V, Z)$$

$$\sum_{i=1}^{n} \lambda_\alpha \; \rho_{ui}(x_\alpha - x_\beta, Z) + \sum_{i=1}^{n} \nu_\alpha \rho_u(x_\alpha - x_\beta, Z) + \mu_2 = \rho_{ui}(x_\beta, V, Z)$$

unter die Bedingungen:

$$\sum_{i=1}^{n} \lambda_\alpha = 1$$

$$\sum_{i=1}^{n} \nu_\alpha = 0$$

(David, 1988).

4.7 Studiengelände

Allgemeiner Hintergrund

Das Studien-Gelände ist ein Wohnquartier, das eine Fläche von ca. 17 Hektare einnimmt; es befindet sich in Zentrumsnähe einer der älteren Städte der Niederlande. Zu Beginn dieses Jahrhunderts befand sich im östlichen Teil des Gebietes eine

Baumwollspinnerei. Die benötigte Energie bezog die Spinnerei aus dem betriebseigenen Gaswerk.

Nach der Schließung der Spinnerei wurden am früheren Betriebsstandort Wohnhäuser gebaut. Als im Verlauf der Bauarbeiten in Wegen und Gärten Farbreste vorgefunden wurden, sah sich der Bezirk Kennemerland veranlaßt, die Kontamination untersuchen zu lassen.

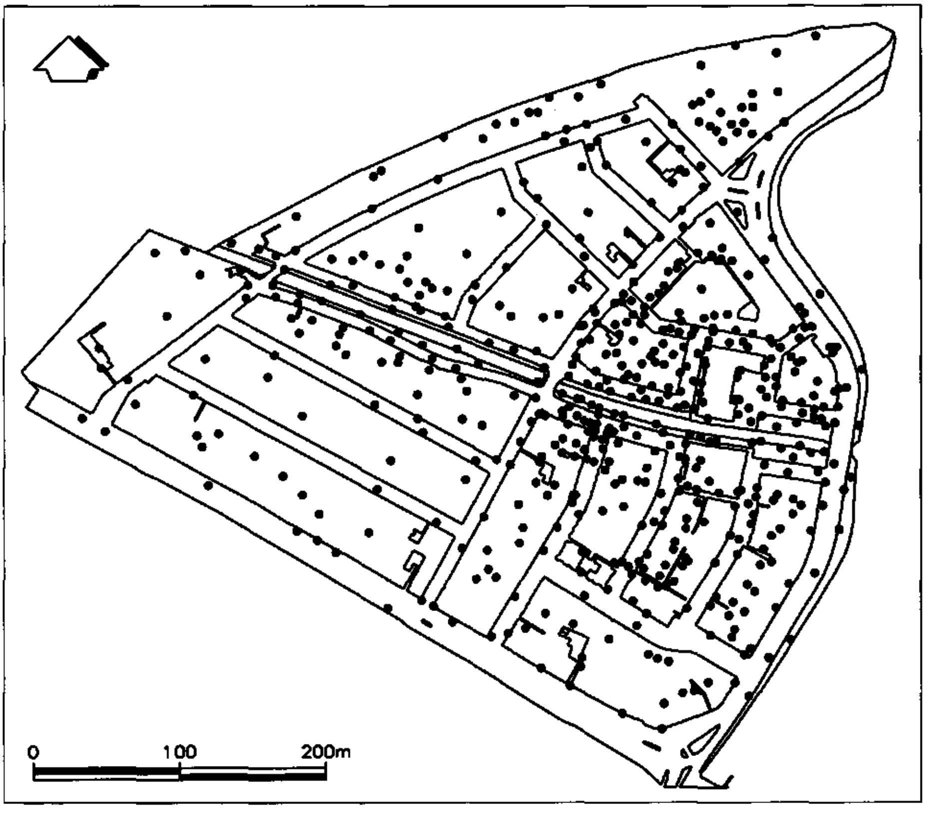

Abbildung 4.6. Lage der Bohrlöcher

Die Farbrückstände entstammen dem Verfahren, mit dem die Baumwolle gefärbt wurde. Chemische Analysen ergaben, daß das Gebiet mit Schwermetallen - vor allem Cadmium und Blei - kontaminiert war; die Untersuchung wurde ausgedehnt. Eine andere Art der Kontamination - mit Polyzyklischen Aromatischen Kohlenwasserstoffen - war durch das betriebseigene Gaswerk verursacht worden. Insgesamt wurden 285 Bohrungen niedergebracht (Abbildung 4.6). Aus Tiefen bis zu 3.5 m unter der Erdoberfläche wurden 719 Bodenproben entnommen. Alle Proben wurden analysiert, so daß ihr Gehalt an Schwermetallen und Polyzyklischen

Aromatischen Kohlenwasserstoffen festgestellt werden konnte. Für die vorliegende Einzelfallstudie wollen wir ausschließlich den Cadmiumgehalt betrachten.

Statistische Charakteristik

Die statistische Analyse, die Ausgangspunkt jeder geostatistischen Studie sein sollte, erlaubt uns einen ersten Eindruck der Variabilität der zu untersuchenden Variablen. Tabelle 4.4 zeigt die wichtigsten statistischen Parameter.

Tabelle 4.4. Statistische Parameter des Cadmiumgehaltes

Anzahl der Proben	Min ($mg.kg^{-1}$)	Max ($mg.kg^{-1}$)	Mittel-Wert ($mg.kg^{-1}$)	Median ($mg.kg^{-1}$)	Standard-deviation	Variations-Koefficienten	Schiefe
719	0.1	460	5.6	1.0	31.4	5.57	9.9

Der hohe Wert des Variationskoeffizienten (5.57) zeigt die sehr hohe Variabilität des Cadmiumgehaltes auf dem Gelände. Es wird ein Höchstwert von 460 $mg.kg^{-1}$ erreicht. Das bedeutet, daß der Boden lokal extrem verunreinigt ist, wodurch eine unverzügliche Bodensanierung unumgänglich ist. Das Frequenz-Histogramm (Abbildung 4.7) zeigt eine sehr asymetrische Verteilung, eine Tatsache, die sich auch im hohen Schiefen-Wert (9.9) ausdrückt. Offensichtlich folgt die Verteilung des Cadmium-Gehaltes keinem gaussischen oder lognormalen Gesetz, was im Fall von Bodenverunreinigungen keine Seltenheit ist.

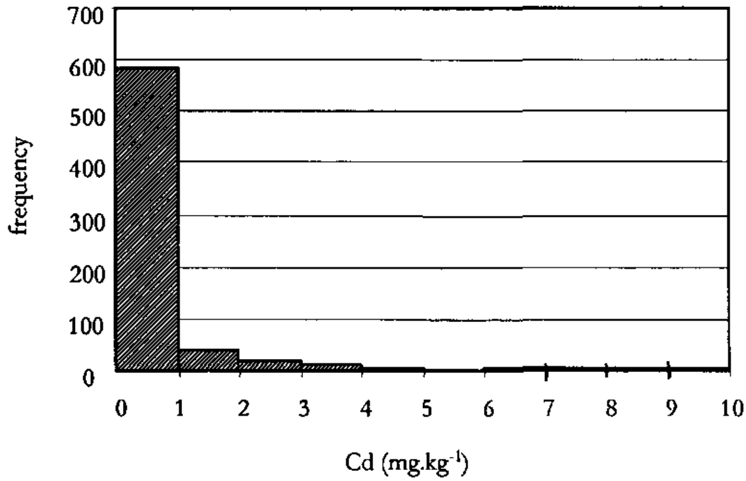

Abbildung 4.7 Häufigkeitsverteilung des Cadmiumgehaltes

Der erste Schritt im Wahrscheinlichkeitskrigen ist das Festlegen einer Anzahl von Cutoffs. Unterschiedliche Cutoffs entstehen durch den Einbezug folgender Randbedingungen:

- Risikoeinschätzung
- technische Möglichkeiten
- wirtschaftlicher Kontext
- politischer Kontext

In diesem Fall sind die Cutoffs auf 2.5 $mg.kg^{-1}$, 5.0 $mg.kg^{-1}$ und 10.0 $mg.kg^{-1}$ Cadmium bestimmt worden. Tabelle 4.5 zeigt eine Reihe von Cutoffs (z_{ci}) und ihre jeweilige Transformationen ($U(z_{ci})$). Diese $U(z_{ci})$-Werte stimmen mit den relativen kumulativen Frequenzen der Distributionsfunktion überein ($F(z(x_i))$).

Tabelle 4.5. Cut-offs z_{ci} und übereinstimmende Transformationen ($U(z_{ci})$)

cut-off ($mg.kg^{-1}$) Z_{ci}	2.5	5.0	10.0
$U(Z_{ci})$	0.873	0.919	0.946

In Tabelle 4.6 sind die Modellparameter der drei Indikatorvariogramme, des Variogramms der umgewandelten Probenwerte und der drei Crossvariogramme.

Tabelle 4.6. Parameter sphärische Modelle verschiedener Variogramme

Variogramm	C_0	C_1, a_1	C
$I_{2.5}$	0.080	0.155,67	0.52
$I_{5.0}$	0.068	0.104,50	0.65
$I_{10.0}$	0.041	0.073,42	0.56
U	0.053	0.084,72	0.63
$UI_{2.5}$	0.029	0.076,63	0.38
$UI_{5.0}$	0.019	0.050,59	0.38
$UI_{10.0}$	0.008	0.034,57	0.23

Diese Modell-Parameter wurden im Rahmen des dreidimensionalen Wahrscheinlichkeitskrigens verwendet, was zu einem dreidimensionalen Raster führte, welches durch die Wahrscheinlichkeit gefüllt ist, daß der fragliche Cutoff überschritten wird.

Abbildung 4.8. Bedingte Wahrscheinlichkeiten, mit denen der Cadmiumgehalt den Schwellwert von 2.5 $mg.kg^{-1}$ überschreitet

Abbildung 4.9. Bedingte Wahrscheinlichkeiten, mit denen der Cadmiumgehalt den Schwellwert von 5.0 $mg.kg^{-1}$ überschreitet

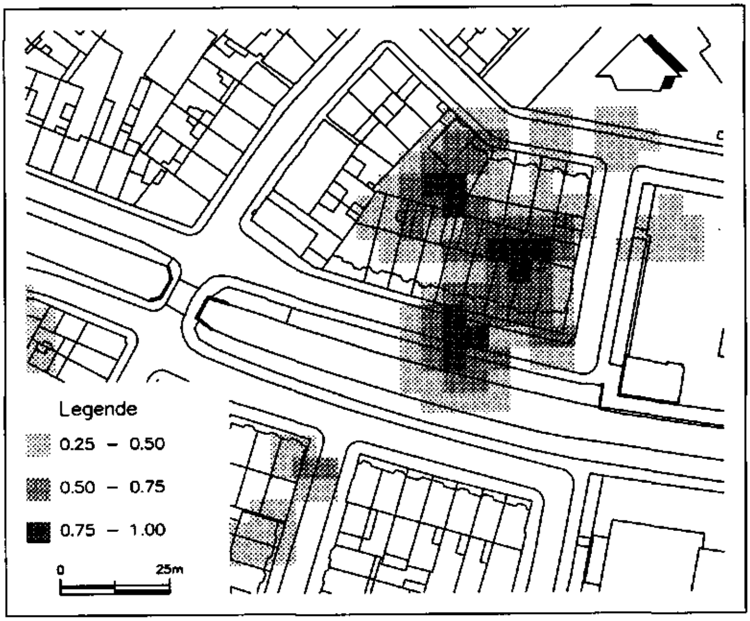

Abbildung 4.10. Bedingte Wahrscheinlichkeiten, mit denen der Cadmiumgehalt den Schwellwert von 10.0 $mg.kg^{-1}$ überschreitet

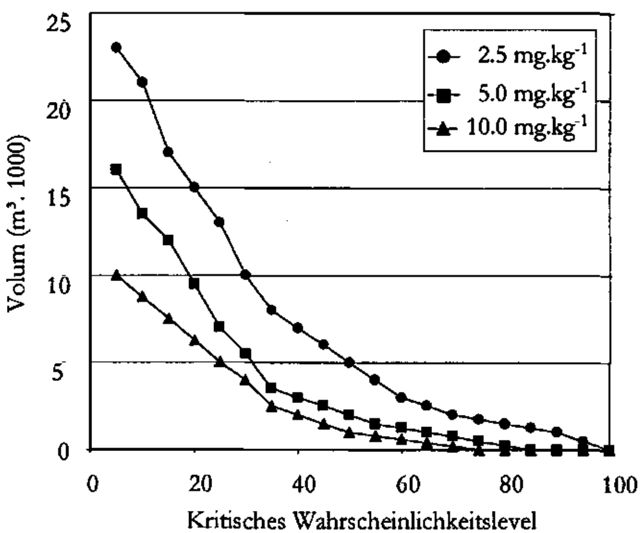

Abbildung 4.11. Volumen an verunreinigtem Boden als Funktion von Cutoff und kritischem Wahrscheinlichkeitslevel

Die Abbildungen 4.8 bis 4.10 zeigen einen zweidimensionalen Ausschnitt der obersten Schicht für drei verschiedene Cutoffs.

Indem wir die Möglichkeiten eines geographischen Informationssystems nutzen, können wir mit Leichtigkeit die Konsequenzen berechnen, d.h. bestimmen, welches Volumen an Bodenmaterial entfernt werden muß, und zwar für jede mögliche Kombination von Cutoffs und kritischen Wahrscheinlichkeitslevels. Abbildung 4.11 zeigt das Resultat einer solchen Berechnung.

4.8 Diskussion

Die Resultate des Wahrscheinlichkeitskrigens können im Rahmen eines Kostenkalkulationsmodells verwendet werden. Traditionellerweise kombinieren diese Modelle einzelne Punkt-Variablen (Volumen, Einheitspreis etc.) miteinander, um den Preis einschätzen zu können. In Tat und Wahrheit kann das Volumen aber größer sein, als es eingeschätzt wurde. Das Wahrscheinlichkeitskrigen liefert auch die Grundelemente für die Entwicklung und Evaluation von Sanierungsszenarien.

Dies kann sowohl für finanzielle als auch logistische Planungszwecke sehr vorteilhaft sein. Wenn es mehrere unsichere Faktoren im Kostenberechnungsmodell gibt, können wir die Monte Carlo Simulation anwenden, um die Kostenfrequenzverteilung zu errechnen.

Chapter 5

AN EXPERT SUPPORT MODEL FOR EX SITU SOIL REMEDIATION

This chapter presents an expert support model recombining knowledge and experiences obtained during ex situ soil remediations. To solve soil remediation problems, an interdisciplinary approach is required. Responsibilities during the soil remediation process, however, are increasingly decentralised, which results in dispersed knowledge. The aim of this model is to optimise knowledge transfer among the various parties involved in contaminated site management. Structured Knowledge Engineering (SKE) has been used as a framework for model development. The model was applied to a petrol pollution underneath a fuel station and to a complex PAH and CN pollution at a former gasworks site. The structured approach requires scrutinising all relevant data in order to answer the questions related to an ex situ soil remediation operation. Moreover, it clarifies the roles of the different disciplines involved in the process.

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... Maar ik ben overtuigd dat ook wij nog zoover van het absolute àf zijn, dat we voelen of bemerken als we een absolute vorm maken deze vorm iets dogmatisch heeft: om kort te gaan, het absolute moet in betrekkelijkheid vooralsnog gebeeld worden ... [6]

5.1 Introduction

When dealing with land use plans, environmental problems are often encountered, mostly caused by contaminated soils. These problems require an interdisciplinary approach, involving land users and owners, scientists - such as toxicologists, economists and engineers - and the decision makers in society (Bouma and Hoosbeek, 1994). As a result of an increasing decentralisation of knowledge and responsibilities many people are involved in solving soil remediation problems. Often, communication has lost much of its efficiency and effectiveness, as has the transfer of knowledge and expertise (Hammer and Champy, 1993). Consequently, the whole process of soil investigation and remediation is at present far from ideal in many ways. Efforts have been made to arrive at a standardised approach (EPA, 1988; Lamé and Bosman, 1993). These efforts are generally aimed at optimisation of national programs and neglect the location specific factors of the individual sites. All of these circumstances - inadequate standardised approaches, shattered responsibilities and lack of communication - are important reasons for the large discrepancy between estimated and actual cost of many remediation projects (Dutch Auditor's Office, 1993).
Many scientists and engineers agree that environmental problems such as soil remediation require an interdisciplinary approach (Verkuijlen, 1989; Salomons and Förstner, 1988; De Groot, 1992). The objective of this study is to better understand the process of ex situ soil remediation design, to identify when and by whom decisions are taken and to identify the necessary amount of data. For that purpose, a start has been made to collect, examine and categorise all existing knowledge and experience on ex situ soil remediations. On the basis of the collected information a decision model was developed, aimed at facilitating communications between the various parties involved in remedial investigations. Moreover, transfer of knowledge will lead to an overall increase of efficiency appraisal. Experiences, which would have been lost, can now serve to expand and develop the soil scientist's knowledge base. In order to develop the model in this chapter the Structured Knowledge Engineering (SKE) (Bolesian, 1991) method has been used. The model was furthermore tested in

[6] Piet Mondriaan in a letter to Theo van Doesburg, 20 november 1915

practice: for a simple petrol pollution underneath a fuel station and for a complex PAH and CN pollution at a former gasworks.

5.2 Methods

Decision-making

The process of soil investigation and remediation is divided into various phases (EPA, 1988; Holtkamp and Gravesteyn, 1993; Gotoh and Udoguchi, 1993). During the first phase, the orientation phase, the necessity for further investigation is determined. During the second, further investigation phase, the remediation urgency is assessed. During the third phase, the remediation investigation, various remedial options are compared. One option is selected and worked out in detail in a remediation plan.

Risk assessment to determine priorities for what should be cleaned up, when it should be cleaned up and how much it should be cleaned up is discussed extensively nowadays. Four key elements are necessary to arrive at an optimal remediation strategy (Blacker and Goodman, 1994). First the risk-goal policy is separated from negotiable technical factors. Second, regulatory policy decisions are translated into measurable criteria to determine the extent of the required cleanup. Third, agreements on critical requirements should be recorded. Finally, technical optimisation is used to develop the most efficient strategy for remediation. In this study we will focus in particular on technical optimisation of ex situ soil remediation. The determining factors, such as soil type and ground water table, are considered as the design criteria.

After reaching an agreement on the requirements, technical and financial factors are decisive for the selection of a technical option. There are several possibilities: the soil can be excavated and treated by several techniques, it can be temporarily stored until better treatment techniques become available or it can be disposed on a landfill site assigned to this purpose.

Knowledge engineering

We will now discuss the development of the decision model. At the heart of a decision model lies the concept of a system. Every system is a construction based on experience (Klir, 1991). Experience, in turn, is expressed as purposeful distinctions either made in the real or in the ideal world, which allows decision models to be distinguished according to the nature of the information. In our study, we focus on real world decision models, for which knowledge and experiences of senior soil remediation experts are used.

Models can also be distinguished according to the purpose for which they are designed. A decision model can present facts and statistics; it can play a supporting role for making decisions or it can be a tool to make interpretations. The model developed by us presents the necessary steps in an ex situ remediation design. To develop such a model we used the model-directed approach "Structured Knowledge Engineering - SKE" which is a standard approach for knowledge system development (Bolesian, 1991).

When building a model according to the SKE-approach several phases are distinguished. The first phase is a preliminary investigation. The objective of this phase is to conceive a knowledge model and a co-operation model, describing how users co-operate with each other and how they co-operate in the future system. The second phase is the initiation. The feasibility of the project and the consequences for an organisation are the subject of this phase. The third phase describes the intended system by integrating the various models created in the preliminary investigation. The fourth phase is the design phase, consisting of the functional implementation and the technical design. During this phase, a distinction is made between system tasks and user tasks. The next phase is the building phase. Since we consider our model as a first prototype which will have to be adjusted in many ways, we present it as a manual and not as a computer model. The last phase is implementation. Once the model has been built, the user needs to be instructed to be able to make full use of its functionality and to avoid unskilful use.

The model is illustrated by means of decision-trees. This was done to show the relation between the initial situation and possible measures. Starting from a specific initial situation and then, following a decision path, one arrives at the exact measure or combination of measures to be taken (Pottjegort, 1992).

5.3 A decision support system

The design of ex situ soil remediation options involves several subprocesses:

1) existing lots of polluted soil have to be identified and characterised;
2) a decision is to be made on the treatment of these lots;
3) if treatments are similar lots should be joined and
4) finally an excavation strategy must be worked out.

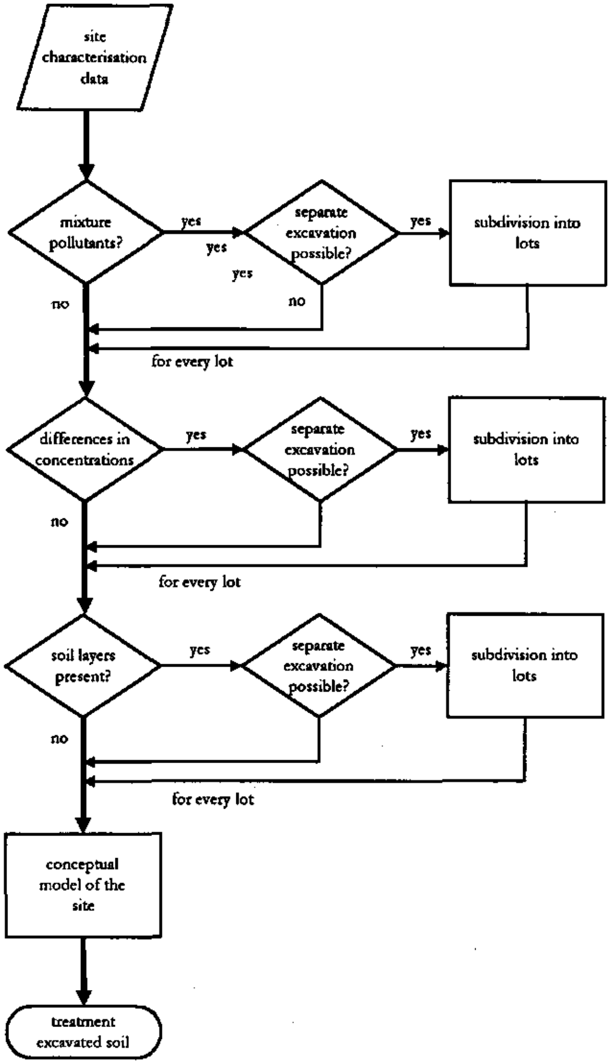

Figure 5.1. Identification of lots

The first process diagram (Figure 5.1) describes identification of the existing lots of polluted soil. Both the heterogeneity of the pollution and the heterogeneity of the soil are considered. Separate excavation of lots polluted with different pollutants may be required to optimise treatment. In a multiple-pollutant case both lots with a single pollutant and lots with multiple pollutants are identified, the latter is a mixture of existing single pollutants. If separate excavation of each lot is possible a decision must be made on the treatment and excavation of each lot. If separate excavation is

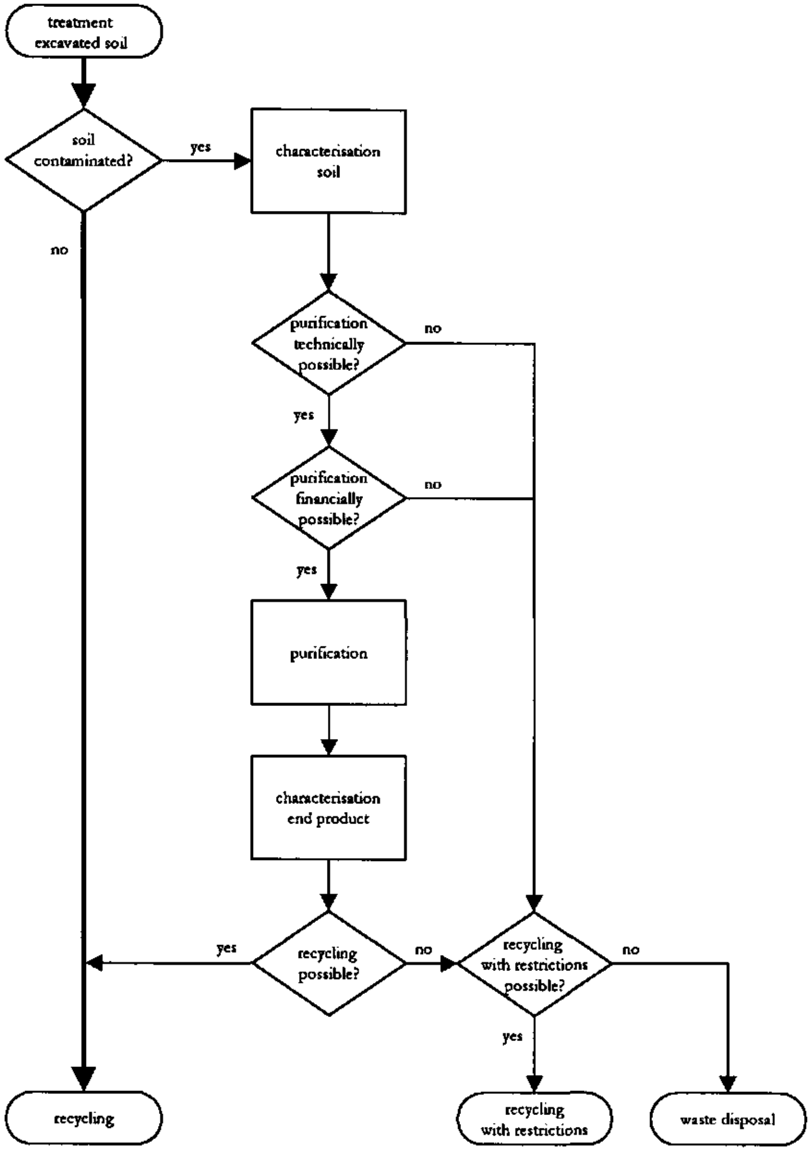

Figure 5.2. Treatment of excavated soil

not possible we automatically arrive at one multiple-pollutant lot. The next question refers to the contamination degree. Sometimes it is possible to distinguish non-contaminated, lightly contaminated, moderately contaminated and heavily contaminated zones. The contamination degree is an important factor for both the treatment and excavation strategy. Again a decision has to be made whether separate excavation of these lots is possible. The next question relates to the heterogeneity of the soil. For future treatment of the soil the question has to be answered whether the soil consists of several soil types and again a decision is to make on separate excavation of these soil texture types. At the end we arrive at a number of different lots characterised by pollutant, level of contamination and soil texture. These lots will be as homogeneous as possible, which will simplify answering the succeeding questions.

The second process diagram deals with the treatment of the soil (Figure 5.2). If the lot is contaminated a further characterisation is needed. The soil type should be specified on the basis of clay % (< 2 μm), silt % (2 - 63 μm), sand % (63 - 2000 μm) and humus % (organic matter) (see Table 5.1). The fraction < 32 to 63 μm is important for cost calculations and for the appropriate treatment method. If this fraction is > 20 % extraction and biological treatment is difficult, yielding thermal treatment as the only alternative, otherwise extraction, biological treatment and thermal treatment are possible.
Debris (bricks, slags, coal particles, etc.) must be separated from the lot by passing the material over a 32 mm sieve, if possible, or over an 80 mm sieve. If separation of debris is impossible this is likely to affect the costs. The variability of the contaminant concentrations may reduce the efficiency of the treatment and should therefore be known. The presence of Hg, EOCl or heavy metals are also important when choosing a particular treatment. In case of a single-pollutant contamination Table 5.1 must be used to determine whether a lot can be treated or not. If treatment is not possible alternatives like re-using the material with restrictions or waste disposal should be considered. In multiple-pollutant cases or if a successful treatment is uncertain (see Table 5.1) experts should be consulted. A process diagram for these cases is not feasible. With growing experience it might be possible to extend the reach of the decision model. If treatment is technically possible, the question arises whether the proposed solutions are financially sound. It should be noted that the answer of this question depends on the chosen policy. After purification the material should be characterised in order to determine whether it can be used with or without restrictions.

Table 5.1. Treatability of polluted soil (SCG, 1995)

Soil texture class	Sandy		Clay		Loam	Peat		
	Clayey sand	Loamy sand	Sandy clay	Loamy clay	Sandy loam	Sandy peat	Clayey peat	Humic material
Clay % (m/m) (<2μm)	0-8	0-8	8-25	8-100	0-25	0-8	0-70	0-30
Silt % (m/m) (2-63μm)	0-17.5	9.5-50	0-42	0-75	50-100	75-85	0-77.5	0-65
Sand % (m/m) (63-2000μm)	82.5-100	50-82.5	50-92	0-75	0-50	75-85	0-77.5	0-65
Organic matter % (m/m)	0-20	0-20	0-25	0-30	0-25	15-25	17-70	35-100

Contamination

Metals Chromium Cobalt Copper Nickel Zinc Arsenic Molybdenum Cadmium Tin Barium Mercury Lead								

PAK Naphtalene Phenantrene Anthracene Fluoranthene Chrysene Benzo(a)anthracene Benzo(a)pyrene Benzo(k)fluoranthene Indeno(1,2,3-cd)pyrene Benzo(g,h,i)perylene PAH total								

non-treatable
expert judgement needed
treatable

The third decision three specifies the excavation strategy (Figure 5.3). If the depth of the excavation is limited (< 1m) or if slope is not steeper then indicated in Table 5.2 no technical measures (TM) are necessary. If else then measures become necessary to assure the stability. The next question addresses the position of the contamination relative to the groundwater table. If the contamination is situated in the unsaturated

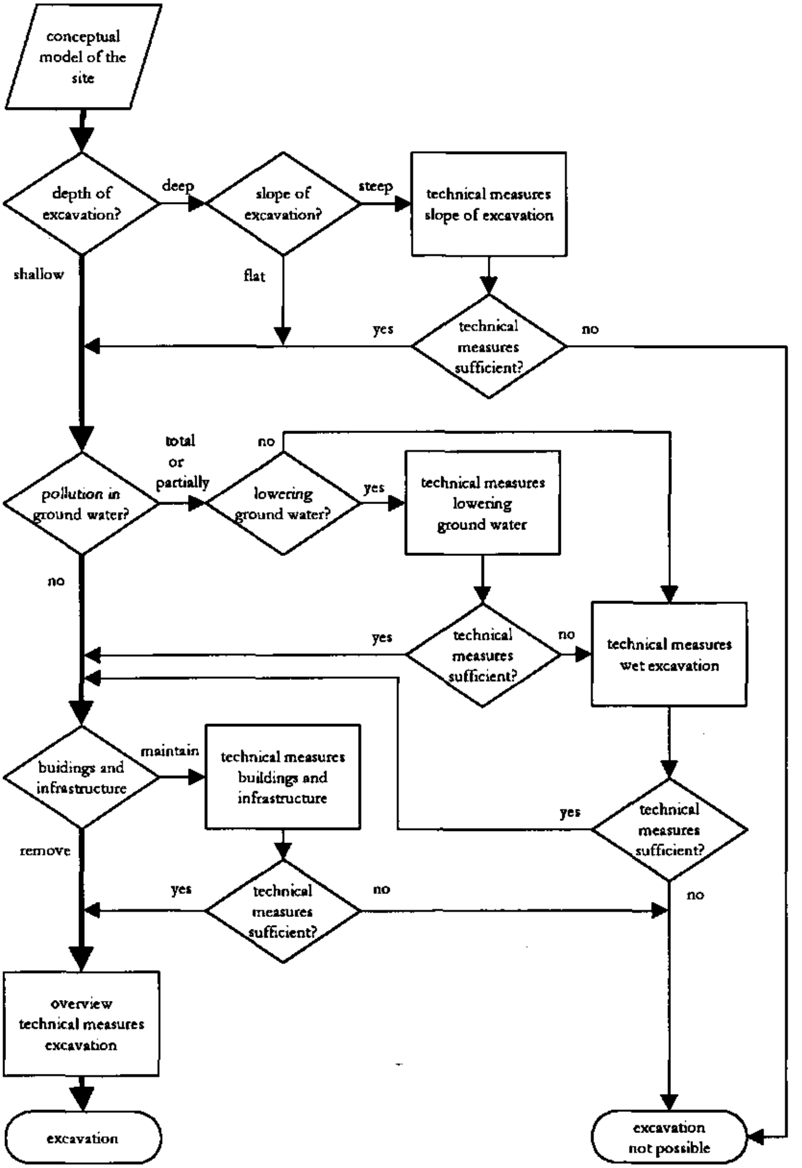

Figure 5.3. Measures related to excavation

zone no problems with the existing groundwater occurs. Otherwise two options are considered: excavation after lowering the groundwater table or excavation below the groundwater table (wet excavation).

Table 5.2. Acceptable slopes

Soil type	Depth below surface	Slope not steeper than
Sand or loam		
Firm, undisturbed	1.00 – 1.50	3.00 : 1
Firm, undisturbed	1.50 – 2.50	1.50 : 1
Firm, undisturbed	2.50 – 4.00	1.25 : 1
Loose, disturbed	1.00 – 4.00	1.00 : 1
Clay		
Very firm, undisturbed	1.00 – 1.50	vertical
Very firm, undisturbed	1.50 – 2.50	2.00 : 1
Very firm, undisturbed	2.50 – 4.00	1.25 : 1
Firm, undisturbed	1.00 – 1.50	Vertical
Firm, undisturbed	1.50 – 2.50	1.50 : 1
Firm, undisturbed	2.50 – 4.00	1.00 : 1
Loose, disturbed	1.00 – 1.50	1.50 : 1
Loose, disturbed	1.50 – 4.00	1.00 : 1

The next question addresses problems related to existing buildings and infrastructure. If the existing buildings will be dismantled prior to the remedial actions no technical measures (TM) will be necessary. However if the buildings are not dismantled and if the foundations and/or the buildings are in bad shape technical measures must be worked out. If necessary roads have to be closed down or rerouting cables and lines may be necessary, leading to more technical measures.

If the worker in charge has answered all the questions all the information relevant for the first conceptual design of the remediation has been processed. Moreover this approach clarifies the position of the different experts and their disciplines in the process. The process rules the disciplines and not the other way around.

5.4 Case studies

A former fuel station

The first study focuses on relatively simple mineral oil pollution in the Netherlands. From 1952 to 1988 a fuel station was situated on a 0.01 ha area. The groundwater table is approximately 3.5 m below the surface. On the basis of an investigation it was concluded that the layer up to 2 m below surface is lightly polluted with polycyclic aromatic hydrocarbons (PAH) and mineral oil. The layer from 2 m till 3.5 m below surface is heavily polluted with mineral oil, petrol and aromatics. The pollution extends to 6 m below the soil surface. Therefore, pollution occurs partly in the saturated zone and partly underneath the existing buildings. The top layer consists of coarse to very coarse (gravely) sand or gravel. Since the site is situated within a groundwater protection zone a multifunctional solution was prescribed.

Table 5.3. The former fuel station lots

Lot	Depth below surface (m)	Characterisation
A	0.0 – 2.0	clean soil
B	2.0 – 3.5	heavily polluted, unsaturated zone
C	3.5 – 6.0	lightly polluted, saturated zone

Petrol is considered as a single-pollutant, although it is a mix of several hydrocarbons (Figure 5.1), because separate excavation of the different hydrocarbons is impossible. A non-contaminated, a lightly contaminated and a heavily contaminated zone are identified (Table 5.3). Separate excavation of the soil for each zone is possible. No distinct soil texture types are identified. Hence it is concluded that the different lots A, B and C differ from each other only in terms of their level of contamination.

Before answering the questions on treatment it is determined whether the lots are contaminated (Figure 5.2). For lot A no treatment is necessary and the soil may be used without limitations whereas lots B and C need to be treated. Separate excavation is necessary to reduce the treatment cost. All lots are classified as sand. The fraction < 32 to 63 μm is less than 20 % and extraction, biological and thermal treatment are possible options. The lots do not contain any debris. The variability of the concentrations of the pollutants in both lot B and C is low and no impact on the efficiency of the treatment is expected. Heavy metals and EOCl were not detected. Table 5.1 indicates that lots B and C can be treated. The next question asks whether the proposed solutions are financially sound. Lightly contaminated soil in Netherlands must be cleaned for less than 45 ECU/ton and heavily contaminated soil for less than 115 ECU/ton. Using thermal treatment as the best available technique this is feasible.

The third process diagram (Figure 5.3) highlights in the excavation strategy. The contamination to be excavated extends to a depth of 6 m below surface and the slope is not steeper then indicated in Table 5.2 (Figure 5.4). Therefore technical measures for stability are necessary (TM 3a). The next question deals with the position of the contamination relative to the groundwater table. If the contamination is situated in the unsaturated zone no problems occur with the existing groundwater. However for contamination in saturated zones two options are considered: excavation after lowering the groundwater table and excavation below the groundwater table. The last questions of the excavation tree deal with problems related to existing buildings and infrastructure. The existing buildings will not be dismantled prior to the remedial actions. The foundations and the buildings require technical measures (TM). During the excavation the roads will be closed down. Rerouting cables and lines are necessary to perform the work, so technical measures are worked out.

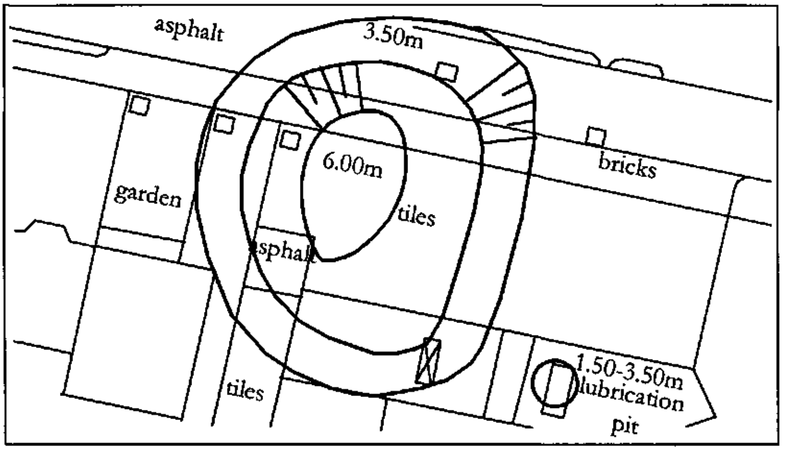

Figure 5.4. Excavation plan for the former fuel station

The application of the decision model to relatively small and simple mineral oil pollution is without problems. The identification of more or less homogeneous lots in relation to the excavation strategy is crucial. Homogeneous lots will not only simplify the answering the questions on future treatment but it will also simplify treatment itself.

The former gasworks Geislingen is situated northeast of the city of Geislingen in Baden-Württemberg in the Federal Republic Germany. The site measures approximately 0.7 ha. The site is bounded on the south by an ICE-embankment (the German high-speed train), on the southeast by a parking-space, on the north by a road and on the northeast by houses. The gasworks was operational from 1890 to 1965 and a number of typical gasworks pollutants (PAH, CN and phenol) have caused a considerable and complex contamination. The site is owned by the city of Geislingen and two private owners. Several buildings are located on the site: some office buildings, two large gasholders, a shed, a storage tank, some cables and remainders of the old gasworks foundation (Figure 5.5). A large part of the site is used as a parking-space. The anthropogenic layer ranges from 0 - 3 m below surface. Underneath this layer the Weißjura is situated (3 - 8 m below surface). The texture of the Weißjura ranges from loam to coarse (gravely) sands and gravel and thus the layer is characterised as an aquifer. The fraction > 32 mm consists of limestone from the Weißjura. Underneath the Weißjura the impermeable Braunjura is situated, which is characterised as an aquitard. For mapping of the contamination the indicator conditioned estimation was used (Zhu and Journel, 1990).

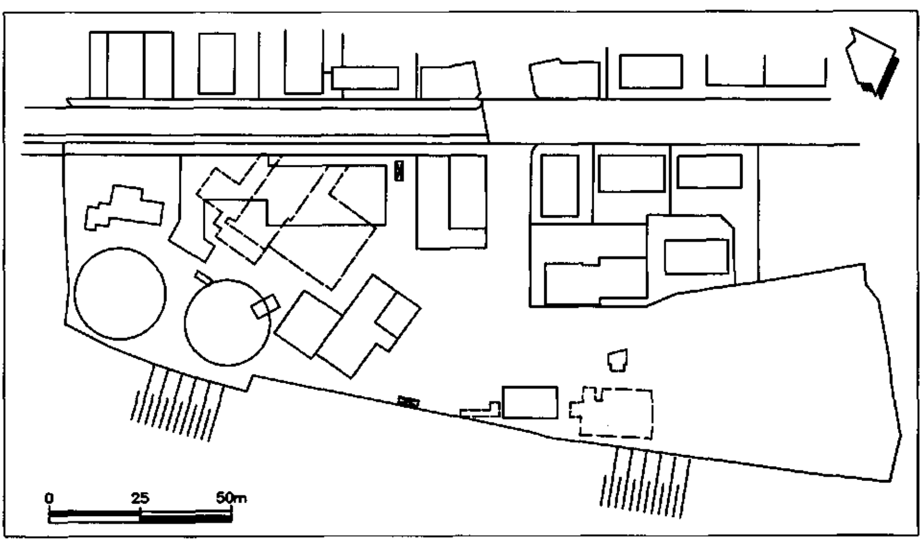

Figure 5.5. Plan of the former gasworks site

A multifunctional solution was not considered, because excavation to a depth of 8 to 11 m below surface alongside the ICE-embankment was not permitted. A possible solution was a partial excavation followed by in situ treatment of the remaining contamination. To arrive at this solution we have followed the process diagrams. The solution involved the excavation of the upper anthropogeneous layer (0 - 3 m below surface) which contains large amounts of debris and the excavation of a number of hot spots (up to 8 - 11 m below surface). Seven lots are identified (Table 5.4). Lots A to D indicate the hot spots, lot F the upper layer and lots E and G are non-contaminated lots which should be excavated in order to reach the contaminated lots.

Table 5.4. Characterisation of the former gasworks lots

Lot	Soiltype	Polluted	Pollutiontype
A	Weißjura	yes	PAH
B	Weißjura	yes	PAH
C	Weißjura	yes	PAH
D	Weißjura	yes	PAH
E	anthropogenic	no	-
F	anthropogenic	yes	PAH, heavy metals
G	Weißjura	no	-

The upper anthropogeneous layer is highly heterogeneous in respect of pollutants, concentrations and debris. PAH, CN, phenols and heavy metals were detected and thus the layer is classified as a multiple-pollutant case (Figure 5.1). Separate excavation within the upper layer however is impossible. Hot spots which can be separated from the upper layer are contaminated with tar oil (mainly PAH). Separate mapping of lighter (2- and 3-rings) and heavier (4- and 5-rings) PAH, shows that the lighter, mobile PAH reaches the deepest contaminated layers. Separate excavation of the different pollutants within the hot spots is however impossible. Both the upper layer and the hot spots contain extremely heterogeneous concentrations. Separate excavation of the lots A to G is possible. The lots (hot spots) A to D are situated within the Weißjura. Below the Weißjura the clayey Braunjura is situated.

Before answering the questions on how to treat this site it should be determined whether the lot is contaminated (Figure 5.2). Lots E and G are not contaminated and therefore no treatment is necessary. Lots A, B, C, D and F are contaminated and treatment is necessary (Table 5.4). Table 5.1 indicates that lots A to D can be treated. Lot F is considered as non-treatable because of heavy metals and other secondary pollutants. Therefore, lot F will be disposed on a suitable waste dump. For lots A to D the large fraction < 32 to 63 μm excludes extraction as a feasible option. Thermal treatment is considered as feasible. For lots A to D the question should be asked whether the proposed solutions are financially sound. The cost for pyrolysis in Germany is approximately 235 ECU/ton, which is acceptable in Germany.

The third process diagram (Figure 5.3) highlights the excavation strategy. The excavations of lot A, B, C, D and F is considered as deep. The slopes of the excavation of lots A, B and C are steep and technical measures are necessary to ensure slope stability. For excavation of lots D and F no technical measures are required. Lots A to C are partly situated in the saturated zone and the groundwater will be lowered locally. Subsidence, effects on the surrounding area, problems related to the discharge of the groundwater are not expected and no technical measures are required. All existing buildings will be dismantled prior to the remedial actions and no technical measures will be necessary. The bitumen covering the parking place will be removed. The majority of the cables and lines will be removed, but for some technical measures are worked out to avoid damage.

Application of the decision model for this complex case worked as well as it did for the simple case. The identification of lots in relation to the excavation strategy, however, is a crucial but a difficult task. In complex cases a quantitative 3D-model of soil and pollution is a necessity. Without such models the successful application of the decision model becomes doubtful.

5.5 Discussion and conclusions

In this chapter we addressed improvement of knowledge transfer and expert's appraisal by an expert support model. Decision-making differs from processing of well-structured intellectual knowledge, analytical reports, abstracted facts and figures, as it is also a matter of personal knowledge and experience and intimate understanding of the business. The system we conceptually developed therefore aims at supporting the expert, rather than replacing him (Okx et al., in press). The decision model is based upon experience within Tauw Milieu b.v., a consultancy processing approximately 1500 soil investigation and remediation projects every year. Testing on two exemplary cases showed that the model forces the user to answer all the crucial questions for an ex situ soil remediation operation and it clarifies the position of the different disciplines in the process.

We performed 3D-visualization of both soil pollution and soil structure since it is the most important starting point for identification of lots of polluted soil. However, the relation between the quality of such a visualisation and the quality of the decisions made within the ex situ soil remediation process is not yet established. Future developments should focus on clarifying these relations, for example by using geostatistics to quantify the uncertainties (Leenaers et al., 1988; Staritsky et al., 1992; Okx et al., 1993). Decision making for treatment of contaminated sites is mainly based on expert judgement and experiences for the existing theory on decision making under risk has not yet been implemented in the practice of soil remediation. Future developments should be aimed at the implementation of the existing theory. Furthermore we expect that separate excavation of identified homogeneous lots will simplify not only the answering of questions on future treatment but it also will simplify treatment itself.

Chapter 6

AN EXPERT SUPPORT MODEL FOR IN SITU SOIL REMEDIATION

This chapter presents an expert support model for in situ soil remediation. It combines knowledge and experiences obtained from previous in situ soil remediations. The aim of this model is to optimise knowledge transfer among the various parties involved in contaminated site management. Structured Knowledge Engineering (SKE) has been used as a framework for model development. This approach requires scrutinising of all relevant data to answer questions related to an in situ soil remediation operation. Moreover, it clarifies the roles of the different involved parties. The approach was applied to a chlorinated hydrocarbon pollution at a dry cleaner's. Use of the expert support model resulted in the development and selection of a new remediation technique.

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Rien ne se perd, rien ne se crée, tout se transforme. [7]

6.1 Introduction

Suppose a dry cleaner's is confronted with a contamination of soil and groundwater with tetrachloroethene (PCE) as result of their washing process. What are the questions that should be answered? What data should be collected? What are the decisions that should be made? They are the kind of questions subject to the development of an expert support model for soil remediation. This study presents an expert support system to answer such questions for in situ soil remediation.

The traditional approach to contaminated land problems is a remedial action aiming at multifunctionality (Robberse and Denneman, 1993). Thus all the functions the soil can possess, given its natural characteristics, are to be re-established. Within the multifunctional framework, in situ techniques aim at reaching threshold values (VROM, 1991 and 1992) in the shortest possible time. This single-perspective view is just one of several possibilities to face the problem. Multi-perspective views, as expressed by Wolf (1993), Selke (1993), van Hattem (1993) and Beinat et al. (1998), include other elements than soil protection and have caused a shift of attention from ex situ remediation to in situ remediation. For example, the triple-perspective REC-framework (Beinat et al., 1998; Okx et al., 1998), takes *risks*, *environmental merits* and *costs* into account simultaneously, and hence aim at optimising a three-fold criterion.

Decision-making, necessary to reach an optimal solution for both frameworks, requires knowledge. Traditional knowledge was general, whereas present knowledge is highly specialised (Drucker, 1994). Consequently, the required interdisciplinary approach to soil remediation yields dispersed and partial knowledge (Okx et al., 1995). In an organisation dispersion or diversification means splintering (Hamel and Prahalad, 1994). It destroys the decision performance capacity of any organisation. If the organisation is composed of specialists with specialised knowledge, the task to perform must be clear, otherwise its members will be confused and will follow their speciality rather than applying their contribution to the common task (Drucker, 1994).

The objective of this study is to clarify the decision making process to the involved specialists by means of an expert support model. The approach is similar to that for ex situ soil remediations (Okx et al., 1995) and to the approach for the design of soil survey schemes (Domburg, 1994). We have collected, examined and categorised knowledge and experiences on in situ remediations described by experts and in the

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[7] After Antoine Laurent Lavoisier (1743-1794)

literature. On this basis a decision model was developed and applied to a PCE-problem in the Dutch city of Breda.

6.2 Soil remediation techniques

During the last decade many techniques have been developed for soil remediation. Two major strategies exist: removal or containment. Successful remediation techniques for ex situ remediation are thermal treatment, extraction/classification and bioremediation, whereas those for in situ remediation include biorestoration, soil washing or extraction and soil venting (Rulkens et al., 1993). Successful containment techniques which prevent contamination to migrate include the application of bentonite screens (Jefferis et al., 1995; Tedd et al., 1995) and soil vapour extraction (Otten et al., 1995). In this chapter we will focus on in situ remediation techniques. Rulkens et al. (1993) distinguishes three major groups of techniques: soil washing, soil venting and biorestoration. Because biorestoration is rather a desired process than a technique, we propose three slightly different groups: soil washing, soil venting and air sparging. Soil washing uses the solubility of a contaminant. By soil washing the contaminant will dissolve in the percolate and by means of a special withdrawal system the percolate is pumped up and treated. Soil venting (Wehrle, 1993) aims at volatilisation and biodegradation of the contaminant in the unsaturated zone. A vapour treatment system will remove the contaminants from the vapour. Air sparging (Gudemann and Hiller, 1988; Eddy et al., 1991; Johnson et al., 1993), an increasingly popular method, involves the injection of air into the saturated zone for the dual purpose of volatilising organic components and enhancing biodegradation (Parsons Engineering Science, Inc., 1995). Overviews of existing technologies are given in Rulkens et al. (1993), Porta et al. (1994), Kovalick and Kingscott (1995), Corver and Versluijs (1995) and Kobus et al. (1996). Detailed information on many remedial techniques can be found in Arendt et al. (1993), Hinchee et al. (1994) and van den Brink et al. (1995).

6.3 Methods

Decision-making

Arriving at the best in situ soil remediation technique involves a decision process. A general model for decision processes (Figure 6.1) is given in Mintzberg et al. (1976). The seven central routines in the figure can be linked to the three main phases of decision-making: problem **identification**, **development** of problem solving alternatives and **selection** of the best alternative. The **identification** phase consists of the central routines: *recognition*, in which the problem is recognised and evokes decisional activity and *diagnosis*, in which the decision makers seek to comprehend the evoking stimuli and determine the cause-effect relations for the decision situation. The **development** phase contains a *search* routine to find ready-made solutions and a *design* routine to develop tailor-made solutions. The **selection** phase contains a *screen* routine to reduce the number of generated ready-made solutions, an *evaluation/choice* routine, which operates in three different modes - judgement, bargaining and analysis - and an *authorisation* routine to obtain approval. The phases and routines can easily be identified in most guidelines for contaminated soil (Gotoh and Udoguchi, 1993; Dreschmann, 1992; Eikelboom and von Meijenfeldt, 1985).

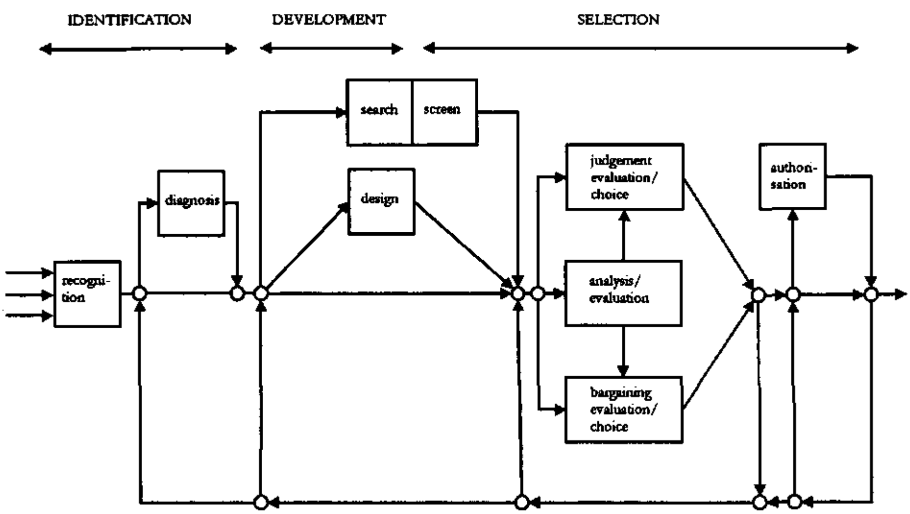

Figure 6.1. A general model for decision processes (Mintzberg et al., 1976)

Interrupts may occur in the process, originating from the decision environment, and can delay, accelerate, stop or restart the decision process. Internal and external interrupts are common in soil remediation and are related to the need and nature of the strategic decision respectively. New option interrupts are less common, but may occur in cases of considerable timelag between authorisation and the final realisation of a project.

Seven types of decision processes according the path taken through the Mintzberg's model are identified (Mintzberg et al., 1976; Nutt, 1984; Janssen, 1992). Only three of these are relevant for soil remediation processes:

1. Basic search decision processes involving development routines consisting of finding the best ready-made solution (Figure 6.2a).
2. Modified search decision processes consisting of finding and modifying ready-made solutions (Figure 6.2b).
3. Dynamic design decision processes, involving complex search and design cycles. These are the most complex decision processes (Figure 6.2c).

Janssen (1992) describes the selection of a technology to clean up a polluted site as a basic search decision process. For in situ remediation techniques, however, ready-made solutions do not exist at present. Therefore, the search and screen routines are always followed by a design routine in which ready-made in situ concepts are modified into solutions. Thus, selection of a remedial technology then should be considered as a modified search decision process, characterised by a development routine in which in situ concepts are modified into tailor-made solutions. In some cases the development routine involves complex search and design cycles and encounters multiple interrupts. This corresponds to a dynamic design process.

Knowledge

Before we will describe engineering a knowledge based decision model, we will give some answers to the question, "What is knowledge?". The inquiry of knowledge has been the central question of philosophy. At one hand rationalism argues that true knowledge is not the product of sensory experience but some ideal mental process. At the other hand empiricism claims that no a priori knowledge exists and that the only source of knowledge is sensory experience. Later philosophers, like Kant and Hegel, attempted at a synthesis between the two streams.
Nonaka and Takeuchi (1995) divide knowledge into explicit knowledge and tacit knowledge. Explicit knowledge can be articulated in formal language including grammatical statements, mathematical expressions, specifications and manuals. This type of knowledge thus can be transmitted across individuals formally and easily. Tacit knowledge, on the other hand, is difficult to articulate with formal language. It is personal knowledge embedded in individual experience and involves intangible factors such as personal belief, perception and value system. Acquisition of explicit knowledge is relatively easy; acquisition of tacit knowledge however requires a more intensive approach. Tacit or personal knowledge can be converted to explicit knowledge through dialogue, discussion, experience sharing, and observation.

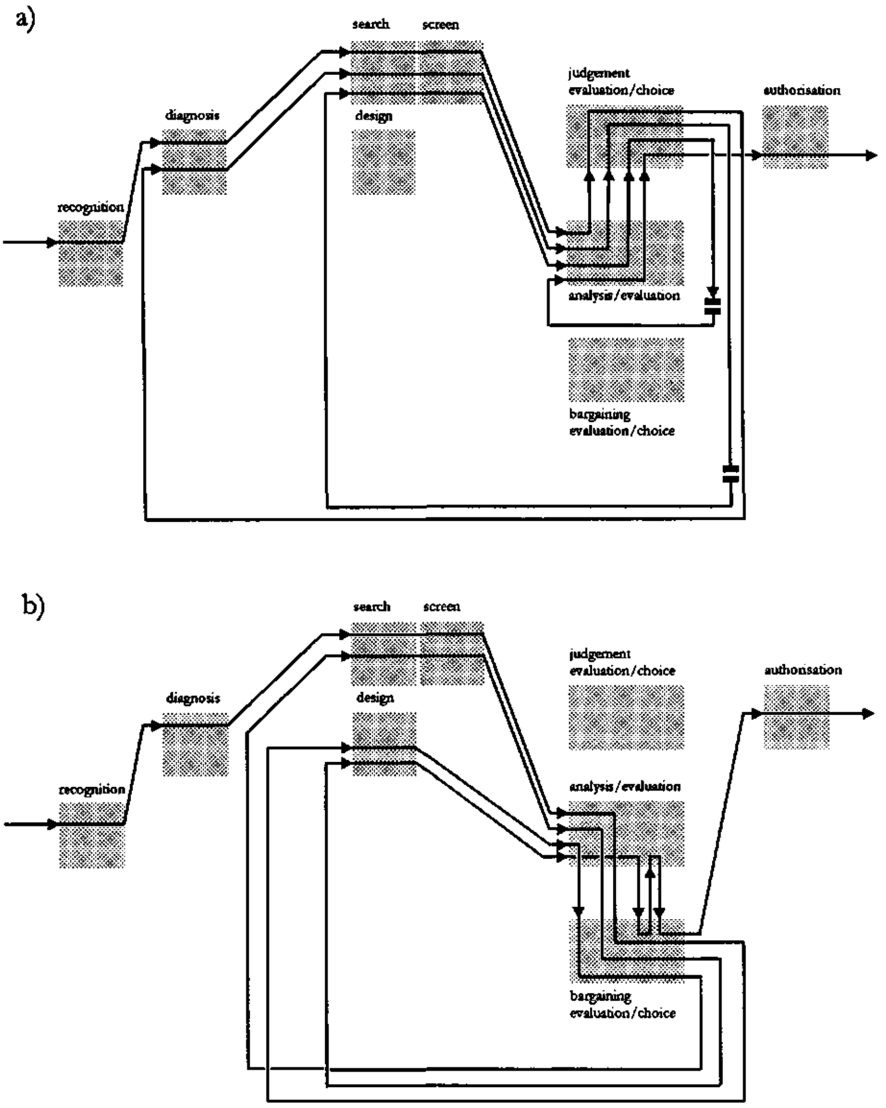

Figure 6.2 a) A basic search decision process
b) A modified search decision process

Following the rationalist tradition, soil remediation experts attain knowledge deductively by applying mental constructs such as deterministic models to predict multiphase flows. Following the empirical tradition they derive knowledge inductively by interpreting sensory experiences such as soil samples or on-line monitoring data. Soil remediation experts, however, do not only use explicit knowledge. Although the experts may not even be aware of it, they also use deep-rooted tacit knowledge like an image of reality (what is or can be) or some vision of the future (what ought to be) and they act accordingly.

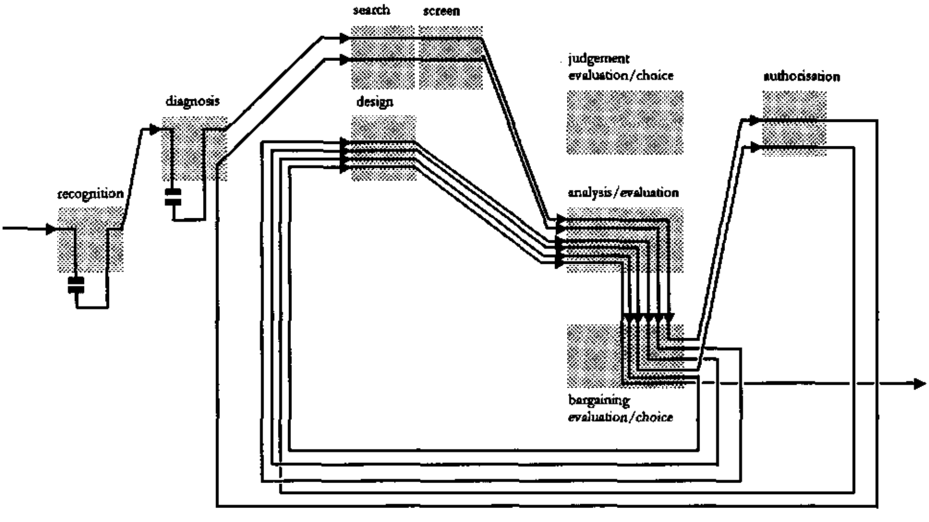

Figure 6.2 c) A dynamic design decision process

Knowledge engineering

For the development of this expert support system we have adapted the framework described by Sol (1990). This framework distinguishes between a **way of thinking**, a **way of modelling**, a **way of working** and a **way of control**.
The **way of thinking** addresses the perspectives from which a problem, such as in situ soil remediation, is considered. Sol (1990) distinguishes the *micro-*, the *meso-* and the *macro-perspective*. From the *micro perspective*, the objective is to improve the performance of an expert. The *meso perspective* aims at improving the performance of an organisation. Thus the co-ordination and the design of the problem solving process is focal. Finally the *macro perspective* concerns the common objective of an interorganisational system and thus aims at improving its performance. In soil remediation such an interorganisational system consists of problem owners, policy makers, universities, consultants and contractors. In this chapter we focus on a synthetic point of view, corresponding to the meso and the macro perspective. Too

much emphasis on the meso and macro perspective, however, obstruct synthesis. Therefore the three perspectives can not be regarded in isolation from each other.

The **way of modelling** refers to making appropriate use of information technology and it usually subdivided into *systelogical*, *infological*, *datalogical* and *technological* problems (Welke, 1977). The *systelogical* problem is addressed when a designer considers the problems an expert solves. It is common to define (part of) the organisation that employs the expert of interest. One can continue by determining and assigning suitable values to its objective and its performance (van Weelderen, 1991). The *infological* problem is addressed when considering which information is processed. The *datalogical* problem is addressed when considering how information is processed and grouped. Finally the *technological* problem is addressed when considering the technology to process information. We used the detailed process schemes of the administrative organisation module of the System Development Workbench of Cap Gemini (1993) to model the workprocess. Such schemes give an overview of actions, documents, archives, choices and decisions related to a workprocess.

The **way of working** covers the phasing, and the methods followed and the techniques applied to construct and represent models. To develop our decision model we have chosen the model-directed approach "Structured Knowledge Engineering - SKE" (Bolesian, 1991) which has its origins in "Knowledge Acquisition Documentation Structuring - KADS" (Schreiber et al., 1988). In SKE the following phases are distinguished: (1) preliminary investigation, (2) initiation, (3) knowledge acquisition, (4) technical design, (5) building, (6) testing and acceptation and (7) maintenance. The objective of (1) is to develop a knowledge model and a co-operation model, describing how users interact and how they will have to interact in the future decision model. The objective of (2) is to address feasibility of the project and consequences for the organisation. In (3) information is collected from the various parties involved in the in situ soil remediation process. During (4) distinction is made between system tasks and user tasks. The fifth phase is the building phase. Once the model has been built, the program needs to be tested and accepted by its users. Finally, since it is safe to assume that anyone with any knowledge will have to acquire new knowledge every four or five years (Drucker, 1994), maintenance of the system is of the greatest importance.

Just as other methodologies SKE, as a **way of control**, addresses efficiency and effectiveness of the design process. After each phase considerations can be made if and how the process should be continued.

6.4 A decision support system

The design of a decision support system for in situ remediation involves five subprocesses:

1) existing lots of polluted soil have to be identified and characterised;
2) a decision is to be made whether in situ treatment of these lots is possible (Figure 6.3);
3) an in situ remedial action technique must be chosen for each lot (Figure 6.4);
4) if techniques are similar lots should be combined and
5) the chosen solution should be worked out in detail (Figure 6.5).

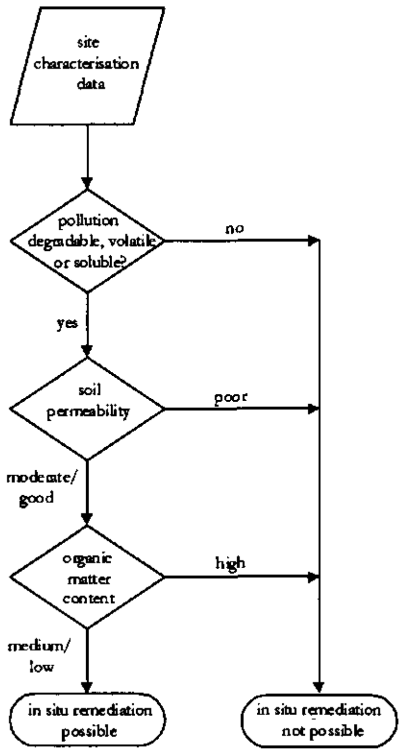

Figure 6.3. Process diagram on the feasibility of in situ treatment

The first subprocess considers the heterogeneity of the pollution and the heterogeneity of the soil. A process diagram for this process is similar to the first step for the design of ex situ soil remediation strategies (Okx et al., 1995). Using the process diagram we arrive at a fixed number of different lots characterised by pollutant, level of contamination and soil texture. These lots will be as homogeneous as possible, which simplifies answering the succeeding questions.

Table 6.1. Contaminant information on in situ treatment

Contamination	Volatility	Biodegradability		Solubility	In situ Possibilities
		Aerobic	Anaerobic		
Hydrocarbons					
Gasoline (C_4 - C_{12})	+	+	-	+	yes
Kerosene (C_6 - C_{15})	±	+	-	+	yes
Gasoil (C_9 - C_{26})	-	±	-	- *	yes
Domestic fuel (C_9 - C_{24})	-	±	-	- *	yes
Lubricants (C_{15} - C_{40})	-	-	-	-	no
Aromatics (BTEX)	+	+	±	+	yes
PAH					
Light (2-3 rings)	±	+	-	± *	yes
Heavy (4-5 rings)	-	-	-	- *	no
Chlorinated Hydrocarbons					
Aliphatic (per, tri)	+	-	+	+	yes
Chlorobenzene	+	+	-	+	yes
Pesticides	-	±	-	-	no
PCB	-	-	-	-	no
Heavy metals	-	-	-	± *	yes

* Solubility can be enhanced by detergents (for hydrocarbons)
or by acidification (heavy metals)

The second subprocess addresses three questions that answers decide whether in situ treatment is feasible (Figure 6.3). The three questions can be answered with Tables 6.1 to 6.3. The Tables are filled with state of the art knowledge and experience on in situ remediations. Table 6.1 answers the question whether contaminants prohibit in situ remediation. Table 6.2 gives an answer to the question whether the soil type is a limiting factor for in situ remediation, just as Table 6.3 answers the question whether the organic matter content is a limiting factor.

Table 6.2. Geohydrological information on in situ treatment

Soil type	K-factor (m/day)	In situ possibilities
Gravel	> 100	Yes
Very coarse sand	10 – 100	Yes
Coarse sand	5 – 10	Yes
Fine sand	0.2 – 5	Yes
Loam	< 0.2	No
Clay	< 0.2	No
Peat	< 0.2	No

Table 6.3. Information related to organic matter content

Organic matter content in %	In situ possibilities
0 - 1 low	Yes
1 - 5 medium	adsorption can give problems
> 5 high	No

The third subprocess addresses the choice of an in situ remediation technique (Figure 6.4), which is strongly related to the question whether the contamination occurs within the saturated zone or not. If the contamination is situated in the saturated zone, presence of light non-aqueous-phase liquids (LNAPL's) should be checked. If LNAPL is present it should be removed before ground water treatment.

In case of volatile or biodegradable LNAPL's venting or sparging are possible alternatives. Else, in case of non-volatile or non-degradable LNAPL's, pumping and possibly treating is an alternative. However, removal LNAPL by pumping is a slow process and feasibility thus depends on the time available. When time is not available, then in situ removal of LNAPL's is not possible and excavation of the layer must be considered. If LNAPL removal is not possible by either of the methods, a successful in situ remediation is unlikely.

When working in the saturated zone without LNAPL, we have to consider lowering the ground water table, because, generally speaking, experience shows that soil venting is a good method of supplying oxygen to the soil. To stimulate the aeration of the soil pores, the soil needs to be unsaturated. Lowering the ground water table increases the efficiency and the speed of the remediation. The choice of lowering the ground water table depends on the following factors:

technical and financial feasibility, required treatment of the ground water, sewer or infiltration capacity and desirability. If lowering of the ground water table is possible and desirable then venting is an appropriate solution, else, we should consider sparging. Finally, if the contamination is not situated in the saturated zone, we can decide to use a washing or venting technique to stimulate volatilisation, solubilisation and/or degradation.

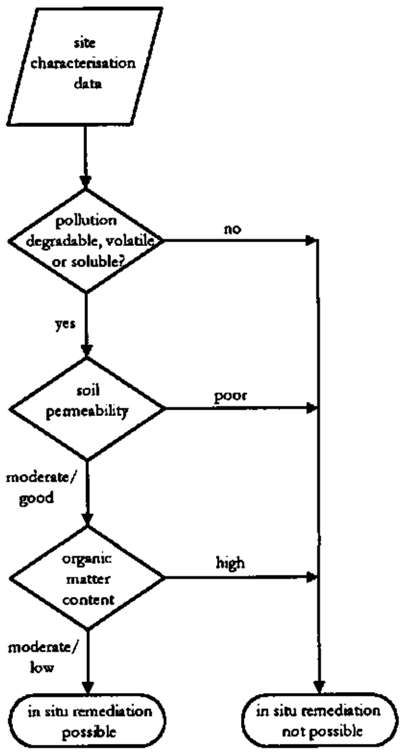

Figure 6.4. Process diagram of the selection of an in situ remediation technique

For the fourth step, combination of different lots, no clear explicit rules emerged from the interviews and therefore no process diagram is given. An experienced soil remediation expert typically performs this step. In this step he has to combine the technical alternatives which emerged from step three into one technically feasible concept.

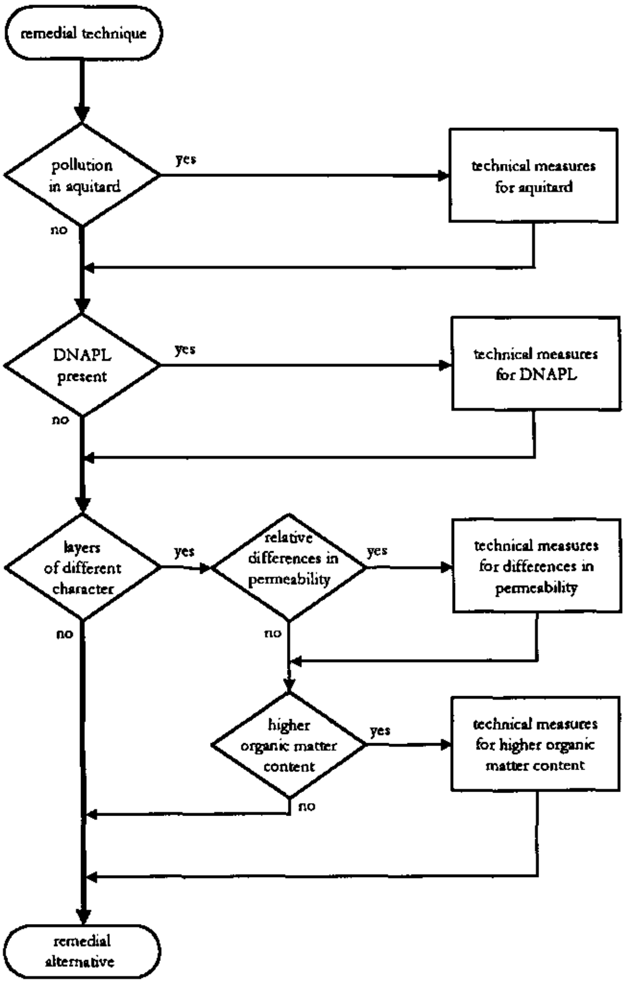

Figure 6.5. Design and engineering tree

The final process diagram (Figure 6.5) addresses step five, design and engineering, of the remedial concept. The first question to answer is whether the contamination in the saturated zone has penetrated any low permeable layers (aquitards). Remediation of these kind of layers is difficult the emission characteristics of low permeable layers cause very long remediation times. Filters on the boundaries of such layers are a possible technical measure. The contamination released by slow diffusion can be captured and removed by pumping. The next question deals with the presence of a dense non-aqueous phase liquid (DNAPL). Chlorinated hydrocarbons or polycyclic aromatics are typical contaminations related to DNAPL's. DNAPL's are hard to remove and in most cases measures will be taken to avoid further spreading of the contaminant. The next question relates to the heterogeneity of the soil. If permeability is varying technical measures will be necessary. Differences in permeability can be caused by differences in texture or by differences in compaction. If a permeable layer contains small layers or lenses with a lower permeability there is the risk of recontamination from these layers and lenses. To enhance vertical flow a number of possibilities exist:

- in case of venting, filters should be set just above the ground water table. Using more than one filter influence the extraction pattern;
- if the position of the layer is known injection of oxygen or nutrients stimulates degradation;
- hydraulic fracturing, to break the lower permeable layers.

If the permeable layer contains layers or lenses with a higher permeability the risk exists of preferential flow pattern, causing the measures to be less effective than predicted. In case of ground water withdrawal the filters must be situated as shallow as possible.

Soil heterogeneity can also be caused by differences in organic matter content. A high organic matter content will decrease the desorption speed, which will cause longer remediation times.

6.5 Case study

Identification of existing lots

A dry cleaner's (approximately 0.2 ha) is situated in the city of Breda in the Netherlands since the early 70's. A contamination of soil and ground water with tetrachloroethene (PCE) as a result of the washing process used till 1993 was detected in 1988. Although the situation was known since 1988 it took almost five years before remedial action was considered and thus the decision process encountered a first internal interrupt. The contamination down to the impermeable layer at a depth of 10 m below soil surface mainly consists of PCE. The soil contains medium to fine sand and the organic matter content is low (<1%). The highest PCE concentration in the groundwater is approximately 50.000 $\mu g.l^{-1}$ (Alphenaar et al., 1996). Although we could identify zones with low PCE concentrations and with high PCE concentrations, separate treatment of these zones is not possible. No distinct soil texture types are identified. Therefore we dealt with a single, relatively homogeneous lot.

Feasibility of in situ remediation

For research purpose the central question of the second process diagram was modified into "Is in situ biodegradation of chloroethenes feasible?". Central in answering this question is Table 6.1, which decides whether PCE prohibits in situ remediation. Although PCE is persistent under aerobic conditions, anaerobic bacteria however are able to dechlorinate it stepwise to trichloroethene (TCE), cis-1,2-dichloroethene (c-DCE) and vinylchloride (VC) to ethene (ETH). Unfortunately dechlorination of c-DCE and VC are the rate-determining steps under anaerobic conditions, and thus these toxic compounds frequently accumulate in practice. In contrast to PCE, the less-chlorinated ethenes (TCE, c-DCE and VC) can be co-metabolically transformed by aerobic bacteria (Hopkins et al., 1993; Alphenaar et al., 1996). Theoretically a two-step biodegradation of chloroethenes should therefore be considered as feasible. Experimental evidence, however, was lacking and to reduce uncertainty we decided to put up a series of laboratory experiments to identify the most suitable (co-) substrates for anaerobic and aerobic degradation. This caused a new option interrupt in the decision process. On the basis of the experiments we concluded that by using the specific metabolic abilities of both anaerobes and co-metabolic aerobes, complete biodegradation of chlorinated ethenes could be obtained. As a result we concluded that complete biodegradation of chloroethenes is possible.

To answer the question whether soil type or organic matter content are limiting factors experimental evidence was again lacking. To answer this question we performed a series of soil-column experiments and thus a second new option interrupt was encountered. On the basis of these experiments we concluded that neither soil type nor organic matter content act as a limiting factor and therefore in situ biodegradation should be considered as feasible.

Choosing a remedial action technique

The contamination is situated in the saturated zone, since chlorinated hydrocarbons are heavier than water the question on floating layers is irrelevant and has to be denied. Lowering the ground water table from the start is not desirable since the remediation should begin with an anaerobic phase. This leads immediately to the conclusion that sparging or washing techniques are suitable for our problem. Since adding carbon- and electron-donors in the anaerobic phase and a co-substrate in the aerobic phase is necessary a washing technique is the most appropriate technique. To establish aerobic conditions in the second phase we decided to apply sparging techniques, for this is a much more local technique than lowering the goundwater. Because the chosen solution is rather unorthodox we decided to discuss the solution with the responsible authorities. Although the authorities had a positive attitude towards the experiment, it caused an external interrupt.

Detailing the chosen solution

On the basis of the results of the previous steps we have chosen a technological concept in which the microbial processes involved are stimulated in spatially separated zones. The PCE contamination present in the spot is anaerobically degraded to TCE and c-DCE. Downstream from this zone, these biodegradation products will be degraded to CO_2 and H_2O in an aerobic environment through co-metabolic processes (Figure 6.6).

Soil and ground water sampling and consequent analysis proved that the contamination has not penetrated the low permeable layer nor did we found any DNAPL's and the contamination is situated in a homogeneous soil. In that context the design and engineering was relatively straight forward, so we could focus on making the biodegradation process work.

The concept basically consists of three aspects (Figure 6.6):

- operation of an anaerobic loop, where extracted ground water ($\pm$ 250 $m^3.day^{-1}$) is infiltrated after addition of methanol and nutrient without any further treatment. In the loop adding methanol stimulates the autogeneous microorganisms. Extended reaction times and intensified contact between bacteria, substrate and contaminants are feasible due to geohydrological isolation of a closed anaerobic loop;
- creation of an aerobic zone downstream of the anaerobic loop, in which the mobile degradation products (mainly TCE, cDCE) migrating from the anaerobic loop are co-metabolically degraded. In this area several infiltration wells are situated where phenol, the co-substrate for the chloroethene degrading micro-organisms, is infiltrated. Four air infiltration filters will create the aerobic environment. The air injection also facilitates the necessary mixing of the substrate and the contamination.
- prevention of spreading of contaminants, products and substrates in the environment. To be sure that none of the infiltrated products spread in the environment and to create a direct flow from the infiltration wells through the system, an extraction well is situated downstream. 150 m^3/d is extracted, one third of which is applied for infiltration of the phenol. The residence time in the aerobic zone is controlled by these flows and must enable complete mineralisation of PCE.

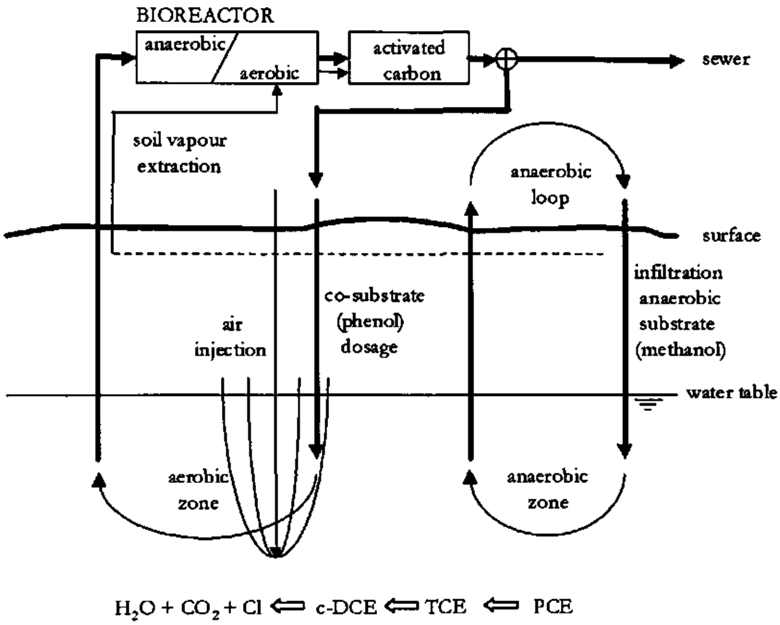

Figure 6.6. Diagram of the planned pilot plant

The soil vapour will be extracted with a continuous flow of approx. 250 m^3/h above the anaerobic and the aerobic sector. If necessary both the extracted ground water and the soil vapour will be treated biologically using a BIOPUR® treatment system. This system was added to the concept in order to obtain the necessary permissions from the authorities. After a final external interrupt permission was given to start the remedial action.

6.6 Discussion and conclusions

In this chapter we addressed improvement of knowledge transfer and appraisal of experts to optimise decision making for in situ soil remediation. Ready-made in situ solutions are barely exist, and hence selection of an in situ technology to clean up a polluted site is not a basic search process. Soil washing, soil venting and air sparging are in situ concepts. Therefore, each time a soil pollution problem is addressed, a search routine selects a concept and a design routine yields a tailor-made solution. Hence, selection of an in situ remedial technology is a modified search process. In some cases the knowledge base is too small for quick decisions and complex search and design cycles are necessary, resulting in many interrupts. In such cases a dynamic design decision process occurs.

The explicit knowledge needed to support decisions was collected during SKE's knowledge acquisition phase. We encountered some reluctance when trying to collect knowledge from the soil remediation experts. This may be caused by our educational system, which focuses on individual knowledge accumulation rather then on developing interpersonal skills needed to disseminate knowledge. To avoid an incomplete knowledge base, the reluctance's were dealt with by making the experts part of the developing team for the expert support model. Although we successfully collected explicit knowledge, the system does not contain all present expert knowledge. Tacit knowledge is not included and therefore the experts and the model are complementary.

The present model is a framework for decision-making processes. It should be noted that the outcome of the experiments described in our case study were added to the knowledge base immediately and will be used the next time a similar problem is encountered. Thus maintenance of the model requires constant attention.

The model as presented here was applied to a chlorinated hydrocarbon pollution at a dry cleaner's. The central question of the second process diagram was modified into "Is in situ biodegradation of chloroethenes feasible?". On the basis of existing knowledge we concluded that theoretically biodegradation of chloroethenes was feasible. A series of experiments showed that the theory was applicable to our problem and thus a washing technique was chosen as the remediation concept. In the following phase the solution was worked out in detail. The test showed that the model forces the user to answer a number of crucial questions necessary for making decisions regarding in situ soil remediation.

Implementation of the model requires continuous cycling through four different modes of knowledge conversion (Nonaka and Takeuchi, 1995). First of all it requires socialisation, the process of sharing experience and creating tacit knowledge. Individuals can acquire tacit knowledge directly from others without using language. Young fieldworkers learn their skills through observation and hence adapt another's individual thinking process. The second mode is externalisation, in which thoughts are formalised into explicit knowledge. Writing this article in order to share our beliefs with others is an example of externalisation. The third mode, combination, is a process of systemising concepts into a knowledge system. It involves combination and reconfiguration of different bodies of explicit knowledge through sorting, adding, combining, and categorising, such as the combination of chemical reactor models with transport models to understand in situ soil remediation. The fourth mode, internalisation, embodies explicit knowledge into tacit knowledge. It is closely related to 'learning by doing'. Once experiences are internalised into individuals' tacit knowledge bases, they become valuable assets (Nonaka and Takeuchi, 1995) and the implementation process can be considered as successful.

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Chapter 7

MULTIOBJECTIVE DECISION MAKING FOR SOIL REMEDIATION PROBLEMS

After deciding whether or not a soil cleanup operation is necessary, the question remains which remedial strategy and technique should be applied. Traditionally, remediation techniques aim at reaching environmental threshold values within the shortest possible time. There is, however, a growing awareness that other aspects should be included when assessing remedial actions. Striving for optimal soil quality at a polluted site may result in the transfer of contamination to other compartments and a considerable use of economic and natural resources. The triple-perspective REC framework simultaneously takes into account risk reduction, environmental performance and costs, and aims at increasing the effectiveness and efficiency of cleanup operations. Within the REC framework, the risk reduction perspective aims at minimising the effects of contamination and remediation on objects at the site. The environmental merit perspective, which stems from an LCI approach, aims at minimising the use of scarce commodities and the contamination of other compartments due to remedial activities. Finally, the cost perspective aims at minimising the total costs in terms of net present value. This chapter shows the method and illustrates an application

This chapter is submitted to:
Int. J. LCA: J.P. Okx, E. Beinat and L. Hordijk

Vóór de Gouden Muur is het een geïmproviseerde janboel, daar krioelt het volk in de luidruchtige chaos van het dagelijks leven, en dat daar niet alles in het honderd loopt is te danken aan de wereld achter de Gouden Muur. Als het oog van de cycloon ligt daar de wereld van de macht, in mysterieuze stilte, beheerst, betrouwbaar, overzichtelijk als een schaakbord: een soort gelouterde wereld van platoonse Ideeën. Dat althans is het beeld dat de machtelozen vóór de Gouden Muur er van hebben. Het wordt bevestigd door de donkere pakken, de geruisloze limousines, de bewaking, het protocol, de perfecte organisatie, de fluwelen rust in de paleizen en ministeries. Maar wie werkelijk achter de Gouden Muur is geweest, zoals u en ik, die weet dat dat alleen schijn is en dat het daar in de besluitvorming net zo'n geïmproviseerde janboel is als er vóór, bij de mensen thuis, op de universiteit, in ziekenhuis of bij bedrijven. [8]

7.1 Introduction

Soil remediation is traditionally concerned with the restoration of soil quality. In the Netherlands, for instance, an almost traditional approach to the remediation of contaminated land is that in which remedial actions aim at multifunctionality, i.e. at reducing concentrations to levels below specific standards (Robberse and Denneman, 1993). The multifunctional framework is based on a single perspective; that is, achieving environmental threshold values within the shortest possible time. There is, however, a growing awareness that other criteria should be included when assessing remediation concepts. One of the reasons for this is that the costs involved in multifunctional operations are no longer politically acceptable. There is also growing recognition that cleanup operations do not necessarily lead to a positive environmental balance. Soil remediation requires the use of resources (such as energy and clean water) and may lead to a net transfer of contamination to other compartments (due to, for instance, air emissions). Therefore, the single perspective implied by the multifunctionality may result in an approach that disregards many relevant concerns for soil remediation.

Decision-making concerning the most suitable remedial action follows a process similar to the mineral exploration and mine valuation process as described in Rendu (1976). This process can be divided into several successive investigation phases. At the beginning of each phase, a decision must be made whether or not to continue the investigation, and if so, which investigation strategy to apply. During each investigation phase, information is obtained on the presence and extent of contamination. As soon as possible a decision has to be made whether or not a remedial action is necessary. If remediation is not necessary, then the investigations

[8] Taken from Harry Mulisch, "De ontdekking van de hemel" (1996), Uitgeverij De Bezige Bij, Amsterdam

can be stopped. If it is necessary, however, the investigations should be focused on the screening of the suitable remedial strategies. The criteria that will influence this last decision are:

- the total impact of remediation strategy on the total risk for humans, ecosystem and infrastructures;
- the total impact of the remediation strategy on scarce commodities, such as soil, groundwater, drinking water, space and energy, and on the quality of the environment as a whole;
- the total impact of the remediation strategy and method on the financial assets of the problem owner (Beinat et al., 1998).

These days, remedial actions are more and more risk driven (ASTM, 1995; CONCAWE, 1997). Risk modelling aims at assessing the risks for humans, ecosystems and physical objects due to exposure to soil contamination. The use of such physico-chemical models as CSOIL (van den Berg, 1991 and 1993; van den Berg and Denneman, 1993), HESP (ECETOC, 1990) and CLEA (Ferguson and Denner, 1993) for human exposure assessment is widespread. Models for ecosystem exposure assessment are available (Van Straalen and Denneman, 1989; Denneman and Van Gestel, 1991), but their use is limited.

The impact of remedial actions on such scarce commodities as soil, groundwater, drinking water, space and energy has not been a real issue and is thus seldom addressed. Although it is generally assumed that a cleanup operation must have an overall positive impact on the environment, some authors (Van der Laar et al., 1997; Volkwein et al., 1997) found evidence to the contrary. In their studies they have used Life Cycle Analysis (Heijungs et al., 1992) to assess all positive and negative effects to the above-mentioned scarce commodities caused by remedial actions.

The impact of remedial actions on the financial assets of the problem owner is generally addressed with detail and precision. Cost estimates are based on robust methodologies and reflect a primary concern of the problem owner: minimising the remediation costs.

The REC framework (Okx et al., 1998; Beinat et al., 1998) takes *risks*, *environmental merits* and *costs* into account simultaneously, and hence aims at optimising a threefold criterion. This article describes the foundations of the approach, shows the main *R*, *E* and *C* components, and illustrates how the framework acts as a decision support tool by means of a case study.

7.2 Decision support

General

In order to develop a decision support tool we need to have a closer look at decision processes. A general model for decision processes by Mintzberg et al. (1976) consists of three main phases of decision making: problem identification, development of problem solving alternatives, and selection of the best alternative. These phases can easily be identified in most guidelines for contaminated soil. Although usually these phases are dynamically linked and do not follow a strict one-way sequence, the process can be conveniently described as a sequential multiphase process. During the first phase, the necessity for further investigation and the urgency for remediation are determined. During the second phase, a number of remedial alternatives are developed. During the third phase (the remediation investigation), various remedial options are compared. Finally, one option is selected and worked out in detail in a remediation plan. This chapter deals with the selection of the best remedial alternative.

Structuring decisions

Creating a decision model requires identification and structuring of the decision objectives within a logical framework and the precise definition of all the objectives considered. The identification and structuring of objectives is the first step in the decision support process. Objectives refer to those issues that matter to the decision maker(s). Simply listing the objectives, however, is not enough: they should be separated into fundamental objectives (the reason why decision makers are interested in the decision situation, such as the minimisation of health effects) and means objectives (the measures to achieve the objectives, such as minimising emissions.) (Clemen, 1996). Fundamental objectives can be organised into hierarchies, with the upper level representing a general objective and objectives at lower levels referring to more specific ones (Keeney, 1992).

A further step is to identify a set of attributes in order to measure the success of a decision in terms of matching the objectives (Beinat, 1997). If a decision involves a single objective (such as reaching threshold values within the shortest possible time), the degree to which the objective is met is often easily identified. With many other decisions (such as those addressed within the REC framework), the choice is made by balancing out multiple conflicting objectives.

The choice of attributes is the result of an interactive refinement process. During the initial stages, attributes are loosely defined. For example, when the REC methodology considers the improvement of soil quality, soil quality plays the role of a decision attribute as it provides a measurement scale on which to express and order decision alternatives. However, how is soil quality measured? In concentrations? In relation to soil quality standards? Such questions have to be answered before the model can be used in practice.

Multiobjective decisions

The majority of decision situations in soil remediation share important similarities. First, decision-makers evaluate a set of **remedial alternatives**, which represent the possible choices. The **objectives** to be achieved drive the design (or screening) of alternatives and determine their overall evaluation. **Attributes** are the measurement rods for the objectives and specify the degree to which each remedial alternative matches the objectives. Finally, **factual** information and **value** judgements jointly establish the overall merits of each option and highlight the best compromise solution (Beinat, 1997). Figure 7.1 summarises the information that plays a role in a multiattribute model. The information items are the multiattribute profiles ($A_1,..,A_m$) allowing measurement of the achievements of the (remedial) alternatives, the value functions (v_i, $i=1,..,n$) representing human judgements, the weights (w_i, $i=1,..,n$), and the multiattribute value function that associates an overall value with each alternative ($v(A_j), j=1,..,n$).

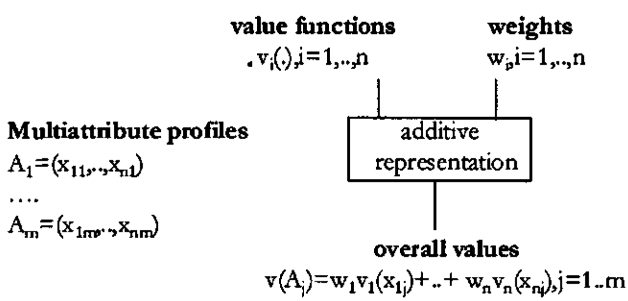

Figure 7.1. Information items in a multiattribute model (Beinat, 1997)

In this example, the overall merit of a decision alternative is computed as a weighted sum of single-attribute performances regarding all attributes. Although this evaluation scheme is very common and widely used, it is important to stress that it can be applied only under very precise conditions. Without going into this topic (see Beinat 1997 for an overview), it is sufficient to say that the additive rule can be applied only if independence conditions across attributes are met. This, in turn, calls for a careful structuring of the decision problems and a careful choice of the attributes.

7.3 The REC framework

General

Decision support systems aim at (Janssen, 1992):

- assisting individuals or groups of individuals in their decision processes;
- supporting rather than replacing the judgements of these individuals; and,
- improving the effectiveness rather than the efficiency of a decision process.

REC is a decision support system using information related to risk reduction, environmental merits, and costs and results (indices *R*, *E* and *C*) for every remedial alternative. Figure 7.2 shows the three perspectives within the framework. Environmental merit concerns quality and scarcity, rather than the actual environmental effects. The following sections describe in some detail procedures to arrive at quantitative values for the REC components.

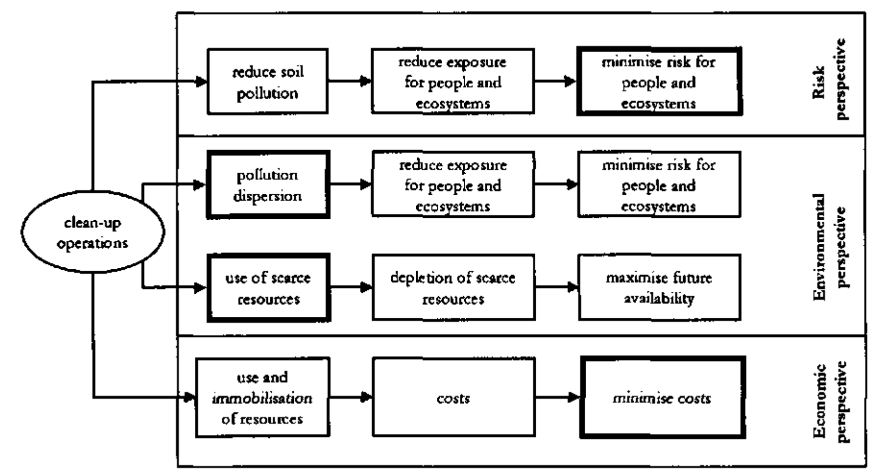

Figure 7.2. The R, E and C perspectives within the framework

Risk reduction

Within the risk reduction perspective, we aim at minimising the negative effects of contamination and remedial action on objects at a specific site. In other words: we aim at maximising risk reduction. This is reflected in the definition of risk as used within the REC framework: Risk: the nature and probability of occurrence of an unwanted, adverse effect on individuals using the contaminated area, local ecosystems and other relevant objects caused by exposure to soil contamination, either directly by the contamination itself or indirectly by the remedial action (CONCAWE, 1997). Risk reduction addresses both the effects of the contamination itself and the effects caused by handling the contaminated material during remedial actions related to objects at a specific site. Both human and ecotoxicity are aspects included in LCA assessment schemes. However, risk assessment tools as CSOIL, HESP and CLEA, are much more dedicated and widely accepted tools for performing this task for soil remediation. The main advantage is that they address risks at a local scale. Although site-oriented approaches within the LCA framework are promoted by some authors (Grieshammer et al., 1991; Fava et al., 1991), there is a lively debate on the actual suitability of LCA for this type of evaluation, and some authors maintain that LCA is not suitable for such an approach (Heijungs et al., 1992). Risk assessment in present soil remediation practices addresses human risks, and thus is predominantly anthropocentric. Within the REC framework, besides human risks, all relevant individuals, species and ecosystems are included in risk assessment, thus giving the process a more biocentric nature (Beltrani, 1997). Once the objects and targets are identified, the effects of contamination exposure x_i should be quantified. In principle there are two strategies which can be followed: direct measurement (e.g. bioassays) or indirect estimates (e.g. samples of contact media) or physico-chemical modelling. Direct measurements provide the most reliable information, but they are expensive and can only be applied to past exposures. Therefore, modelling is the only practical alternative. In the Netherlands, CSOIL and HESP are widely accepted models, and are used within the REC framework. However, the framework is suitable for and can accommodate other risk assessment models.

From the assessment models we obtain the calculated exposure (x_i) for each of the possible remedial actions $i=0,1,..,n$ where 0 signifies a maintaining of the status quo and $1,..,n$ the remedial alternatives under evaluation. If d signifies a toxicological threshold and m the number of exposed objects, then

$$y_i = mx_i/d$$

signifies the normalised exposure for remedial alternative i. For the calculation of the normalised exposure for humans, the Total Daily Intake (TDI) is used as the toxicological threshold value, whereas for ecosystems the concentration where 50% of the species are at risk (HC50) is the adequate threshold. When the risk index for humans is 1, then the level equals the Maximum Tolerable Risk (MTR). When the risk index for the ecosystem is 1, then 50% of the species is expected to be affected. Once the normalised exposure is determined for all relevant objects, the total normalised exposure can be calculated by simply adding the different normalised exposures. If the time is represented by t, then the total risk R for remedial alternative *i* can be calculated by

$$R_i = y_i\, t$$

After that the risk reduction can be calculated. It is defined as:

$$R_0 - R_i = y_0\, t - y_i\, t$$

In cases where exposure also takes place during remedial action, time is being subdivided in time before the remedial action t_0, time during remedial action t_d, and time after remedial action t_i. The reason why the time before the remediation is considered is that some techniques have an immediate impact, while others show a delayed effect. Thus, risk reduction in that case can be defined as

$$R_0 - R_i = \frac{y_0(t_0 + t_d + t_i) - (y_0 t_0 + y_d t_d + y_i t_i)}{t_0 + t_d + t_i}$$

This formula can be applied to each remedial alternative *i*. From a risk reduction perspective, the higher the score, the better the performance of the alternative.

Environmental merit

Risk reduction only addresses the adverse effects on individuals, local ecosystems and other relevant objects. Therefore we turn to the concept of environmental merit (EM) through which we can include environmental costs and benefits beyond those encompassed by the risk assessment. A cleanup operation may result in the use of scarce resources (e.g. energy), transfer the pollution to other compartments (e.g. emissions to surface water during operations), and to secondary effects (e.g. the emission of greenhouse gasses due to combustion of fossil fuels).
The evaluation of cleanup operations in terms of environmental merit is based on an Environmental Merit Index (EMI). This index is constructed by rating the performances of cleanup options against a list of measurable aspects and by aggregating these performances with a weighting scheme. The main steps for

constructing an EMI are based on a traditional multicriteria analysis (cf. Beinat, 1997) and can be described as follows:

1. Select a list of measurable variables that determine the environmental quality of a remedial option.
2. Quantify the performances of each remedial alternative for each of these aspects. The results are organised in a Performance Table.
3. Establish a normalisation function for each aspect. This serves to transform performance scores into a comparable scale. The results are organised into a Normalised Performance Table.
4. Establish the weights for each aspect. Weights represent the relative contribution of each aspect to the EMI. Intuitively, a weight indicates how important an aspect is compared to another.
5. Calculate the weighted sum of the normalised scores resulting in the EMI index for an alternative.

The aspects which are considered in environmental merit are a cross section of the typical Life-Cycle Inventory aspects (ISO, 1996) and of the specific aspects relevant to soil remediation. The reasons for going beyond the usual LCI indications can be summarised in two points:

1. LCI applied to soil remediation does not cover all aspects considered relevant by soil remediation practitioners. The amount of space taken up by the remedial actions, for instance, is considered a relevant decision factor in soil management, especially where space is a scarce resource.
2. The LCI provides a list of impacts with a strong emphasis on global effects (such as acidification, eutrophication, global warming, etc.). In soil remediation, not only global effects but also regional and local considerations are important. This calls for a more balanced selection of evaluation criteria.

An analysis of the practice of LCI and soil remediation and interviews with expert panels led to the selection of the aspects listed in Table 7.1. After quantification of the performances of the alternatives as regards the evaluation aspects, we obtain a performance table (see Table 7.5). Improvement of soil and groundwater quality, prevention of groundwater pollution and emission to surface water are expressed in equivalents of a general form

$$X(eq) = \sum_j \int_{c=t_j}^{c_{max}} \left(\frac{V_{b,j}(c) - V_{e,j}(c)}{N_j} \right) dc$$

where,	$X(eq)$ =	improvement of soil or groundwater quality, prevention of groundwater pollution or emission to surface water
	j =	contaminant j

V	=	volume of soil, groundwater or surface water with concentration c_j in m^3
c_j	=	concentration of contaminant j in mg/m^3
s_j	=	target value of contaminant j in mg/m^3
e	=	end concentration
b	=	begin concentration
max	=	maximum concentration
N_j	=	normalisation value for contaminant j

In this equation the removed freight of a contaminant j above a fixed target value s_j is calculated. The target value refers to a good soil, groundwater or surface water quality. After that the difference between begin and end situation is divided by the normalisation value N_j to obtain soil, groundwater or surface water equivalents in m^3. The normalisation value is some value above which the quality is expected to be problematic. Finally the contributions of the different contaminants j are summed.

Table 7.1. The evaluation aspects for environmental merit

Aspects	Units
Positive outcomes	
Improvement soil quality	G(eq) [m^3]
Improvement groundwater quality	W(eq)[m^3]
Prevention groundwater pollution	T(eq) [m^3]
Negative outcomes	
Soil use	[m^3]
Groundwater use	[m^3]
Energy use	[J]
Air emission	[ton]
Emissions to surface water	O(eq) [m^3]
Final waste	[m^3]
Occupied space	[m^2 x year]

Usually it is impossible to identify the best option on the basis of the performance table. Normalisation and weighting are the steps necessary to reach such a conclusion. Normalisation serves to transform each score into a normalised, dimensionless score. To perform normalisation it is necessary to select normalisation curves and the normalisation range. The range, for each aspect, serves to specify two anchor points to which we attach the reference points of the normalised scale, usually the 0 and 1 values. In REC, these scores are linked to the zero alternative (i.e. the status quo) and to a reference case representing an average cleanup operation. Replacing the reference case by the maximum cleanup operation yielded resolution problems in cases where the maximum case differed too much from the rest of the alternatives. However, it is important to note that the choice of these anchor points is arbitrary and that other anchor points could be equally suitable.

The normalisation rules in REC are linear functions. Examples of normalisation curves are given in Figure 7.3. As an example, the energy curve attributes to each energy consumption a value score between 0 and a negative value. The value of -1 is attached to the reference score selected. If a remedial alternative does not consume fossil fuels, its normalised score will be 0. The higher the consumption, the more negative the normalised score. Normalised scores are usually organised into a table similar to the performance table. The difference is that the normalised scores now represent the degree to which each aspect contributes to the environmental merit of each alternative. The higher the normalised score on a given aspect, the better the performance of that alternative on this aspect.

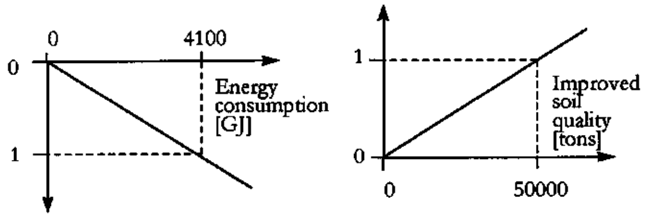

Figure 7.3. Example of value functions

The overall quality of a remedial option is a weighted combination of the normalised scores. Intuitively, weights represent the relative importance of one attribute compared to another. The higher the weight attached to an aspect, the more the results of this aspect are driving the evaluation. Weights are assessed through interviews. Precise 'question and answer' protocols are used to ensure that the respondent provides weights that are a true representation of his/her decision strategy. It is very important to note that weights give an indication to the following question: "How much would you give up in a variable to achieve a given improvement on another?". Therefore, weights can be considered as exchange rates between aspects. The interpretation of weight as a concept of importance or priority is not sufficient in this context. We do not ask people simply "Which criteria is more important?" but "How much do you want to trade-off between criteria?". This distinction is something far beyond a purely academic consideration. It actually distinguishes between an intuitive estimate of importance (linked to general perceptions, feelings and attitudes of a person) and a precise statement of the decision strategy to be applied in practice.

Weights should be assessed by those who have the power to evaluate the alternatives and to fix priorities for evaluation, and whose job is to do so. Environmental merit deals with non-local and public aspects. Thus, this power resides in some supra-local authority, for instance the provinces. Weights can be different for different situations. Within a certain area, for instance, high weights can be given to groundwater consumption. This may reflect a policy for that area, which aims at minimising groundwater use. Consequently, it is necessary to test the variability of weights for different conditions. In REC, a panel of experts, who have been individually interviewed, assesses the weights. The panel includes experts working for the provinces, city councils and the Ministry of Environment, and also those at the level of large companies. Table 7.2 shows the average weights obtained during nine interviews with Dutch experts. Note that the sum of the weights should always be 1.

Table 7.2. Average weights

Aspects	**Weights**
Positive outcomes	
Improvement soil quality	0.11
Improvement groundwater quality	0.08
Prevention groundwater pollution	0.20
Negative outcomes	
Soil use	0.09
Groundwater use	0.19
Energy use	0.06
Air emission	0.05
Emissions to surface water	0.11
Final waste	0.07
Occupied space	0.04

The weighted sum of the normalised scores provides the EMI results, which can be used to rank the remedial options from the best to the worst in terms of the environmental merit perspective.

Costs

An important issue when evaluating remedial alternatives is the total expected costs. However, there is more than one reason to promote the joint presentation of risk reduction, environmental merit and costs. The first reason is that decision making without cost indication is not very likely. The second is that the relation between the three perspectives may contain important information for policy makers. If, for instance, the correlation between cost and environmental merit is positive (low cost/low environmental merit and high cost/high environmental merit) then the problem owners will not automatically choose for the most sustainable remedial alternative. In this case, regulations would be useful. On the other hand, if the correlation is negative (low cost/high environmental merit and high cost/low environmental merit), then problem owners will be most happy to choose sustainable alternatives, and regulations become unnecessary. Within the REC framework costs are defined as *"Cost: the amount of financial sacrifices made to enable remedial action."* Cost estimates in the Netherlands are performed using the cost categories as defined in the Soil Protection Guidelines (VROM, 1997). However, other guidelines could also be used within the framework. The Soil Protection Guidelines use the following categories:

- basic costs, e.g. preparatory work, demolition work, control system(s), supervision;
- continuous costs, e.g. maintenance, post-closure measures
- replacement costs;
- overheads;
- other costs, e.g. damage compensation paid to third parties.

Cost calculations have to be made for each of the remedial alternatives. The basic cost figures are produced using design and engineering tools, such as the knowledge-based decision model for ex situ soil remediation (Okx et al., 1995) or the biosparging and bioventing expert support system (Okx et al., 1996). Since non-exhaustive data sets are used to estimate cost-determining factors (such as the volume of polluted soil, the level of contamination, and the duration of the remediation), the results of these calculations are of an uncertain nature (Okx et al., 1993a and 1993b; de Wit et al., 1995). Thus, the costs are best represented by an expected value (E), a standard deviation (s) and a risk avoidance factor (k). If we consider the expected value normally distributed with mean E and standard deviation s, then there will be a constant area between the mean and an ordinate k that signifies a given distance from the mean in terms of standard deviation units. In other words the larger the risk avoidance factor k, the smaller the risk that the costs will be higher than the given value of a remedial alternative:

$$v_i = E_i + k \cdot s_i$$

where, v_i = value of remedial alternative i

E_i = expected cost of remedial alternative i

k = risk avoidance factor

s_i = standard deviation of the expected cost of alternative i

Some alternatives involve actions in which money has to be spent within a relatively short period of time, while others involve actions in which the spending of money is distributed over a long period.

In order to compare these alternatives, the costs should be expressed as net present value (NPV):

$$NPV = \sum_{t=1}^{n} \frac{CF(t)}{(1+i)^t}$$

where, NPV = net present value
$CF(t)$ = cash flow in year t
i = interest rate

Thus the framework yields net present values for all alternatives. From a cost perspective, the lowest value signifies the best alternative.

Integration

The choice between cleanup options is a multiobjective problem. Ideally, the alternative that maximises the risk reduction, maximises the environmental merit and minimises the costs would be chosen. In practice, there is no single alternative that is better than the others in all respects are. Therefore, the decision has to be made by weighing the pros and cons of each alternative. It is important to realise that REC provides the main information support for this purpose. The REC indices:

- describe the main consequences of the cleanup process in a simple and direct way;
- introduce structure to a complex decision problem;
- provide decision-makers with a picture of the situation that brings the complexity of the decision within manageable proportions.

REC provides this support in the form of three indices: risk reduction, environmental merit and costs. Nevertheless, selecting the best alternative depends on many issues. However, three particular items are relevant for understanding the role of REC in the decision process:

- The degree to which REC covers all relevant concerns in the decision process. For instance, in a specific decision context it may be necessary to include the reactions of local inhabitants to noise or disturbance in addition to the REC indications. This may highlight some alternatives and disfavour others beyond the indications of REC.

- The decision rule or, in other words, the relative importance of *R*, *E* and *C* expressed in weighting factors. Note that these weighting factors refer to weighting between *R*, *E* and *C*, whereas those mentioned in the environmental merit section refer to weighting within *E*. Examples of decision rules are:
  - select the project with the highest risk reduction and the highest environmental merit score within the available budget of 10 million guilders;
  - select the most risk-efficient project (the one with the highest ratio between risk reduction and cost), provided it has a positive merit score;
  - select the cheapest project, provided it has a significant risk reduction;
  - select the project with the highest weighted sum of the three REC indices;
- The degree to which the evaluation is the result of a formal analysis (such as the application of the decision rules mentioned above) or of negotiation and compromise between the actors in the decision process (such as the owners of the site, the authorities involved, etc.).

7.4 Case study

Site description and remedial alternatives

The case study concerns a former dry-cleaner in the city of Almelo, the Netherlands. The contamination of soil and groundwater with chlorinated hydrocarbons (predominantly cis-1,2-dichloroethene (c-DCE)) is a result of the chemicals used in the dry-cleaning process. Besides the chlorinated hydrocarbon contamination, two hot spots contaminated with mineral oil resulting from an unknown process were found. The maximum concentration of c-DCE in the ground is 120 $mg.kg^{-1}$, whereas its maximum concentration in the groundwater is 30000 $\mu g.l^{-1}$. The maximum concentration of mineral oil in the ground is 40000 $mg.kg^{-1}$, whereas its maximum concentration in the groundwater is 950 $\mu g.l^{-1}$.

Three remedial alternatives are suitable for the clean-up the site:
1. Alternative I: a multifunctional alternative with excavation of the contaminated ground and groundwater treatment;
2. Alternative II: an isolation (IBC) alternative in which further spreading of the contamination is prevented;
3. Alternative III: an in situ alternative combined with a complete groundwater remediation.

For the calculation of the normalised exposure for humans and ecosystems, the Total Daily Intake (TDI) and the concentration where 50% of the species are at risk (HC50) were used. The results of these calculations are given in Table7.3. The risk reduction scores of the multifunctional and the in situ alternative are almost identical. The slightly lower score of the in situ alternative is due to the longer duration of the remediation and the somewhat higher end concentrations. The risk reduction of the isolation (IBC) alternative is significantly lower.

Table 7.3. Risk reduction (*R*), environmental merit (*E*) and costs (*C*) in million ECU of three different remedial alternatives

	Alternative	*R*	*E*	*C*
I	Multifunctional alternative	1.00	1.68	3.7
II	IBC (isolation) alternative	0.60	-0.20	1.0
III	In situ alternative	0.96	0.57	1.4

The results of the environmental merit calculations are given in Table 7.4. Although improvement of soil quality, energy use and occupied space are the highest discriminating criteria, all aspects (except prevention of groundwater pollution and of emissions to surface water) have some discriminating power between alternatives. After normalisation and applying the weights from Table 7.2, we obtain the overall environmental merit score (Table 7.3).

Table 7.4. The evaluation aspects for environmental merit and the performance table

Aspects	Units	MF	IBC	In situ
Positive outcomes				
Improvement soil quality	G(eq) [m^3] (x1000)	990	0	378
Improvement groundwater quality	W(eq) [m^3] (x1000)	526	227	527
Prevention groundwater pollution	T(eq) [m^3] (x1000)	0.62	0.62	0.62
Negative outcomes				
Soil use	[m^3]	189	0	25
Groundwater use	[m^3] (x1000)	1600	1344	1800
Energy use	[GJ]	5822	2091	3141
Air emission	[ton]	85	31	46
Emissions to surface water	O(eq) [m^3]	0	0	0
Final waste	[m^3]	650	0	25
Occupied space	[m^2 x year]	501	900	300

The cost of the alternatives given in Table 7.3 is net present values for a period of 30 years and is expressed in ECU.

7.5 Discussing the decision

General

The REC framework provides three indices: risk reduction, environmental merit, and costs, as shown in Table 7.3. The risk index indicates that the MF alternative has the best score in terms of risk reduction, and is followed closely by the in situ alternative; the isolation (IBC) alternative performs considerably worse. The environmental merit index confirms the best score for MF. However, the in situ alternative performs substantially worse on the M index. The isolation (IBC) alternative acquires a negative score, which in environmental merit terms means that IBC is worse than the status quo. Thus, MF is best for both risk reduction and environmental merit. The cost of this alternative, however, is by far the highest. The in situ alternative is less expensive, but the least expensive is the IBC option. The overall situation can thus be described by means of a ranking table as presented in Table 7.5.

Table 7.5. REC ranking table

	R	*E*	*C*
MF	1	1	3
IBC	3	3	1
In situ	2	2	2

Since there is no overall best option, we cannot make a straightforward decision. The decision depends on the weights the decision-maker attaches to the three perspectives.

In other words, the overall quality of an alternative within a specific decision context is a function of the REC indices and of other factors besides REC. This function can in turn be explicitly or implicitly used for negotiations between actors. In formulas:

$$Q(A) = f\,(R^A, E^A, C^A, \text{Other factors})$$

where $Q(A)$ is the quality of alternative A; R^A, E^A, C^A are the REC indices for that alternative; and f is the decision rule, explicitly or implicitly used. On this basis, there are several possible approaches to the decision on the basis of REC outcomes. They can be broadly classified as shown in Table 7.6.

Table 7.6. Possible uses of the REC indices

	REC outcomes are sufficient to make a decision	Other factors contribute to the decision
The decision rule is explicitly known as a function of R, E and C.	**(1)** The alternatives are evaluated by applying the decision rule and are ranked from best to worst.	**(2)** The alternatives are evaluated by applying the decision rule and are ranked from best to worst. The decision rule can be extended to include other aspects (e.g. noise nuisance to the surroundings) or these aspects can be added to the REC results as additional information.
The decision rule is not made explicit.	**(3)** The evaluation of alternatives is based on REC outputs, but the pros and cons are discussed between decision-makers and the decision is reached through bargaining and negotiation.	**(4)** The evaluation of alternatives is based on REC outputs and on additional factors, but the pros and cons are discussed between decision-makers and the decision is reached through bargaining and negotiation.

The distinction between these four cases is in practice less sharp than depicted in Table 7.6. For instance, part of the decision rule is explicit (e.g. maximise the risk reduction/cost efficiency) while leaving other factors to the negotiation (e.g. the role of environmental merit). The following considerations can support the decision making process when deciding on which approach to apply in a given case:

Situation 1

This is the case when the REC results are sufficient to cover the consequences of cleanup and there is a clear necessity to make the decision as explicit and transparent as possible. By making the decision rule explicit it is also possible to implement a precise decision policy, for instance, by stating the weights of the *R*, *E* and *C* results. It also allows for the comparison of decisions made in different cases.

As shown above, there are several decision rules that can be applied. The choice depends to a large extent on the specific decision context. As an example, if *the* R, *E* and *C* indices are considered to provide independent indications of the quality of the alternatives, then a weighted sum of R, *E* and *C* could be appropriate. This would mean, first, the normalisation of R, *E* and *C* indices into a comparable scale, and then the assessment of their relative weights. The decision rule will look like:

$$REC^A = rR^A + eE^A - cC^A$$

where *r*, *e* and *c* are the weights attached to the R, *E* and *C* indices.

Situation 2

In this case, the situation is similar to that in situation 1. Again, the main reason for this will be the necessity of making the decision as explicit and transparent as possible. The additional aspects to be included can be treated in the same way as the REC indices, and thus be explicitly included in the decision rule. However, they can also be taken into account without explicit inclusion in the decision rule.

Situation 3

There may be good reasons to avoid selecting an explicit decision rule and to leave the decision to the negotiation between the actors. R, *E* and *C* provide the main source of information, leaving it up to the actors to make an analysis and thus the final choice. Typical situations where this can happen are when:

- alternatives are very different and one or few distinctly emerge. In this case, it may be unnecessary to proceed to a formal evaluation;
- several actors are involved, and they have different perspectives on the decision rule to be applied. In this case it may be difficult to agree on a single decision rule, and it may instead be convenient to negotiate directly on the choice to be made;
- viewpoints of the actors are very different and the search for an explicit decision rule may enlarge these differences. Skipping the search for explicit decision rules may make room for negotiations and conflict resolution.

Situation 4

This case is similar to situation 3 above, the only difference being that here additional aspects play a role. The same comments can be made also in this case. The negotiation process, because there are more issues to argue, will necessarily be more complex. However, this also leaves scope for more options to be included in the negotiation, for instance by including compensation measures.

If it is considered that the REC results are sufficient to cover the consequences of the cleanup and the parties involved agree on explicit and transparent decision making (which corresponds to situation 1 in Table 7.6), then the overall quality of the decision alternatives can be computed on the basis of an explicit combination of the REC indices.

Table 7.7. Example of three sets of weights for *R*, *E* and *C* and the corresponding best option

Weight sets	*r*	*E*	*C*	Alternative
1-1-1	0.33	0.33	0.33	MF
REC	0.20	0.30	0.50	In situ
RC	0.10	0.00	0.90	IBC

As an example, the application of the additive combination leads to the results shown in Figure 7.4 for the different sets of weights given in Table 7.7.

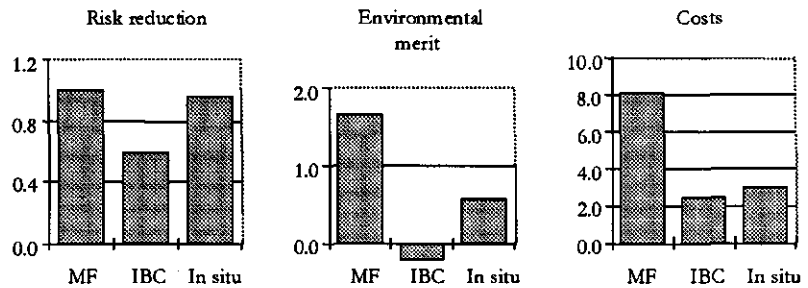

Figure 7.4. Illustration of the formal integration of *R*, *E* and *C*

7.6 Conclusions

Decision-makers in the field of soil remediation are normally confronted with a number of remedial alternatives representing the possible choices. The REC framework shows that the theory of decision analysis can be put into practice and help support the selection of the best cleanup option for a remedial site. The REC framework provides evidence concerning the risk reduction, environmental merit and costs of each alternative through three indices: the *R*, *E* and *C* indices. Ideally, the alternative that maximises the risk reduction, maximises the environmental merit and minimises the costs would be chosen. In practice, however, there is no single alternative that is better than the others in all respects are. The decision on which option to select, therefore, has to weigh the pros and cons of each alternative.

Selecting the best alternative depends on many things, but three items in particular are relevant to the understanding of the role of REC in the decision process:

- the degree to which REC covers all relevant concerns in the decision process;
- the decision rule: the relative importance of *R*, *E* and *C* expressed in weighting factors ;
- the degree to which the evaluation is the result of a formal analysis - such as the application of the decision rules above - or of negotiation and compromise between the actors in the decision process.

Current tests being run by soil consultants in real application settings show that the REC framework is relatively easy to apply and that it substantially improves the transparency of decision-making processes.

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Chapter 8

CONCLUSIONS

Er is een traditie dat Zenkloosters alleen onderdak verschaffen aan zwervende Zenmonniken als zij een koan [9] *hebben opgelost. Zo zou eens een monnik aan een kloosterpoort geklopt hebben. De monnik die open deed groette niet maar zei "Toon me het gezicht dat je had voor je ouders geboren waren." De monnik die onderdak wilde, glimlachte, trok de sandaal van zijn rechtervoet en sloeg zijn ondervrager er mee in het gezicht. Daarop werd de poort geopend en hij werd vriendelijk ontvangen. Na het eten raakten gastheer en gast in gesprek en de gastheer complimenteerde zijn gast met het prachtige antwoord.*

"Weet je zelf het antwoord op de koan die je me gaf?" vroeg de gast.

"Nee" zei de gastheer "maar ik wist dat het antwoord dat jij gaf goed was. Het was spontaan. Je aarzelde geen moment. Het klopte precies met alles dat ik over Zen gehoord en gelezen heb."

De gast zweeg en dronk thee.

Plotseling kreeg de gastheer argwaan. Er was iets in het gezicht van de gast dat hem niet beviel.

"Jij kent het antwoord toch wel he?" vroeg hij.

De gast begon te lachen en liet zich achterover vallen van plezier. "Welnee" zei hij, "maar ik heb ook veel over Zen gehoord en gelezen". [10]

8.1 Objectives revisited

Before the final conclusions are presented, I would like to revisit the objectives of this thesis as presented in Chapter 1. In summary, the aim of this thesis was to foster soil remediation research towards a fully-fledged problem-oriented discipline in order to yield efficient and effective solutions for soil pollution problems. Two core objectives were derived from this aim. The first objective was to supply soil remediation research with a paradigmatic framework. The second was to facilitate consistent problem analysis and decision-making.

Soil pollution problems are often investigated from different disciplinary or analytical perspectives (see Chapter 2). Thus, we have to deal with a myriad of paradigms and difficulties in achieving progress in soil remediation research are believed to stem from the lack of a single paradigm. System science is said to provide such a paradigm. The question, however, is whether this 'new paradigm' is replacing the paradigms of the contributing disciplines or not. In my opinion it does not and it should not. Instead the 'new paradigm' offers a unifying principle

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[9] Ko-an (Jap.; Chin.: Kung-an), anekdote, verhaal, dialoog, in het zenboeddhisme ontwikkeld als methode om langs directe, intuïtieve weg inzicht op te wekken.

[10] Taken from Janwillem van de Wetering, "De lege spiegel" (1971), De Driehoek, Amsterdam

that refines the paradigms of the contributing disciplines rather then replacing them and as such the term 'paradigmatic framework' should be preferred to 'paradigm'.

The benefits of a systems approach in terms of facilitation of consistent problem analysis and decision-making are demonstrated in three critical phases in the soil remediation process:
1. Decisions regarding the identification or characterisation of the soil pollution problem in Chapters 3 and 4;
2. Decisions regarding the development of ex and in situ soil remediation concepts in Chapters 5 and 6;
3. Decisions regarding the selection of a remediation alternative in Chapter 7.

Although I have chosen for a systems approach, it should be noted that some disciplines are more supportive than others are in a particular phase. Statistics (Chapter 3) and spatial statistics (Chapter 4) do play an important role in the identification phase of the soil remediation process, whereas soil science plays a dominant role in the development phase (Chapters 5 and 6). Finally, decision-making is the issue in the selection phase (Chapter 7).

8.2 Decisions on characterisation

Statistical decision theory

Statistical decision theory, rather than (spatial) probability theory, provides the answer when trying to evaluate investigation strategies for soil remediation problems. The following steps are to be taken into account:
- Definition of the problem. Without identification and definition of the problem(s), no valuing is possible;
- Listing of options. At least two courses of actions must be available, and they must be unequal in ability to achieve the goal;
- Definition of criteria. Some measure of expected value must be specified;
- Analysis of the options. Each possible course of action must now be carefully studied in terms of desired outcomes;
- Choice of a course of action. A decision is made when a particular course of action is chosen from among those available.

Working with decision trees is rather straightforward and gives insight in the possible courses of action or decision strategies. Decision trees allow careful reflection on specific probability and value inputs. To evaluate conditional decisions one can make use of Bayesian reasoning.

In a number of examples I have shown that (cheap) minimal sampling does not always lead to the lowest costs or highest payoff. Valuing sampling schemes or experimental set-ups proves to be useful. In this article value is expressed in ECU, however, in the literature value is often referred to as a measurement of relative liking or preference on the part of a decision maker for particular outcomes. I have assumed a linear relation between ECU and value. This is not always a realistic assumption. Decision-makers can have three different attitudes towards risk: risk avoiding, neutral or risk seeking. These attitudes can be expressed in the form of preference or utility functions. A conversion to utility functions is possible but requires additional assessments and is outside the scope of this thesis.

Estimating probabilities

Statistical and geostatistical methods are important in solving the valuation problem, since they provide the probabilities to calculate the expected value or costs of a sampling strategy. Moreover, typical problems such as the determination of the presence of a hot spot, the estimation of a concentration or the mapping of locations where concentrations exceed environmental thresholds can only be solved when the proper (geo) statistical tools are applied.

I have shown in two cases – a former pigment factory and a former cotton mill - that probability kriging can yield important inputs for cost calculation models as they give information on the probabilities of volumes above environmental thresholds.

8.3 Decisions on the development of ex and in situ soil remediation concepts

Ex situ remediation

I have addressed improvement of knowledge transfer and expert's appraisal by a knowledge-based decision model for ex situ remediation. Decision-making differs from processing of well-structured intellectual knowledge, analytical reports, abstracted facts and figures, as it is also a matter of personal knowledge and experience and intimate understanding of the business. The system I conceptually developed aims at supporting the expert, rather than replacing him. The decision model is based upon literature and experience. Testing two exemplary cases showed that the model forces the user to answer all the crucial questions for an ex situ soil remediation operation and it clarifies the position of the different disciplines in the process and, finally, it yields sound remedial alternatives.

I recommend 3D-visualisation of both soil pollution and soil structure since it is the most important starting point for identification of more or less homogeneous volumes of polluted soil. Furthermore I expect that separate excavation of identified homogeneous lots will simplify not only the answering of questions on future treatment but it also will simplify treatment itself, for heterogeneous material is difficult to treat.

In situ remediation

The explicit knowledge needed to support decisions was collected during the knowledge acquisition phase. I have encountered some reluctance when trying to collect knowledge from the soil remediation experts. This may be caused by our educational system, which focuses on individual knowledge accumulation rather then on dissemination of knowledge. Although I successfully collected explicit knowledge, the system does not contain all expert knowledge. Tacit knowledge is not included and therefore experts and model are complementary.
The present model is a framework for in situ remediation design. The outcome of the experiments described in my case study were added to the knowledge base immediately and will be used the next time a similar problem is encountered. Thus maintenance of the model requires constant attention.

The model as presented was applied to chlorinated hydrocarbon pollution at a dry cleaner's. The central question of the second process diagram was modified into "Is in situ biodegradation of chloroethenes feasible?". A series of experiments showed that the theory was applicable to our problem and thus a washing technique was chosen as the remediation concept. In the following phase the solution was worked out in detail. The test showed that the model forces the user to answer a number of crucial questions necessary for making decisions regarding in situ soil remediation.

8.4 Multiobjective decision making for soil remediation

Ex and in situ remediation design is mainly about identification and development of problem solving alternatives. The final choice is similar to the selection phase of

Mintzberg's decision model. This phase consists of a *screening* routine to reduce the number of generated ready-made solutions, an *evaluation/choice* routine, which operates in three different modes - judgement, bargaining and analysis - and an *authorisation* routine to obtain approval.

Decision-makers in soil remediation have to choose between different remedial alternatives. The presented REC framework puts decision theory into practice and supports selection of the best option. The REC framework provides evidence concerning risk reduction, environmental merit and costs of each alternative through the R, E and C indices. Ideally, the alternative that maximises the risk reduction, maximises the environmental merit and minimises costs would be chosen. In practice, however, there is no single alternative that is better than the others in all respects are. The decision on which option to select, therefore, has to weigh the pros and cons of each alternative. Selecting the best alternative depends to a large extent on the following three items:

- the degree to which REC covers all relevant concerns in the decision process;
- the decision rule: the relative importance of R, E and C expressed in weighting factors ;
- the degree to which the evaluation is the result of a formal analysis - such as the application of the decision rules above - or of negotiation and compromise between the actors in the decision process.

Current tests being run by soil consultants in real application settings show that the REC framework is relatively easy to apply and that it substantially improves the transparency of decision-making processes.

8.5 Future developments

General

This study aimed at providing expert support for soil remediation problems. Although it should be concluded that the study succeeded in doing so, further study is needed to refine and extend the support. The developments are classified according to the three main phases of decision making:

- Developments within the identification phase;
- Developments within the development phase;
- Developments within the selection phase and beyond.

Developments within the identification phase

I have recommended 3D-visualisation of both soil pollution and soil structure being a good starting point for identification of lots of polluted soil. Spatial analysis tools, such as geographical information systems, support evaluation of this kind of spatial information. Identification of lots or classification in more or less homogeneous units is essential for understanding and solving the problem. As I can use spatial analysis tools for 3D-modelling and –visualisation, I could also use these tools for classification. Aiming at a goal-oriented system would involve the development of a set of anticipating classification rules. All remedial techniques work within margins and it would be highly recommendable to use these margins as classification rules, thus these margins need to be identified. Since performance of a remedial technique depends on several (dependent) properties, multivariate analysis for classification is needed. Since margins are not likely to represent sharp boundaries between applicable and not applicable, the use of fuzzy logic instead of Boolean logic should be considered.

In many earth science studies, the size of each sample is important (Isaaks and Srivastava, 1989). There is a relation between the size or "support" of our data and the distribution of their values and their interpretation. I can imagine that using standard soil samples to predict the feasibility of a remedial technique, would result in much variability between these predictions and thus a lot of uncertainty regarding the decisions to make. One sample might lead to a "not feasible", whereas another might lead to "feasible". If I would use a pilot plant on a semi-technical scale, then more stable predictions could be expected. The mixing of high and low values, that is to be expected with a pilot plant, would give us less erratic values.
The issue is to find out whether small-volume samples to predict (remedial) processes at a larger scale could be used (Stein, 1997).

Developments within the development phase

Development of technologies for treatment of contaminated sites is mainly based on expert judgement and experiences. Future developments should be aimed at trying to anticipate on the final decision, and, thus at trying to use decision criteria from the selection phase during the development itself. Ideally, the development of alternatives should be aimed at maximising risk reduction, maximising environmental merit and minimising the costs. Thus, this type design bears characteristics of Constructive Technology Assessment (Daey Ouwens et al. 1987). In order to improve current design practices a set of rules which links design actions or design modules to the decision criteria should be provided. Once such design practices are implemented, the number of infeasible designs is expected to decrease, which in its turn will lead to increased efficiency and effectiveness.

Developments within the selection phase and beyond

The relation between the quality of 3D-visualisation of pollution and soil and the quality of the decisions made within the soil remediation process is not yet established. Future developments should focus on clarifying these relations. If the resources in terms of time and money are limited, but improvement of the visualisation quality is aimed at, then it is necessary to know exactly what data are required to make these improvements. Some decisions are sensitive to minor changes in input values and some others are not. Sensitivity analysis could be an answer to establish the relation between input uncertainty and the output quality. The objective is to offer the decision-maker a possibility of exploring the effect of different input.

A tool for continuous improvement is the Plan-Do-Check-Act (PDCA) cycle originating from Shewhart (1986) but commonly seen in relation to Deming (1986).
This cycle results in improvement because one first plans the activities (setting the norm), then carries them out, checks the result (verification) and then improves any deficiencies or areas that may require improvement. I found no evidence of attempts to close the Deming cycle in either a review concerning the contaminated land policies in some industrialised countries (Visser, 1993) or in the standardisation literature.
This cycle or loop of cause-effect relations is a 'feedback process'. Without feedback there can be no improvement and the same mistakes will be made over and over again. The 'check-routine' should therefore be considered as crucial and should be an integral part of all remedial operations. The results of remedial actions should undergo evaluation to see how it compares with prior expectations. If they do not match expectations, then actions should be reconsidered.

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Samenvatting

Menselijke activiteiten hebben tot gevolg gehad dat op veel plaatsen grond en grondwater zijn verontreinigd. In de jaren tachtig is men zich – mede door toedoen van wat we toen nog liefkozend milieuactivisten noemden – de omvang van de problematiek ten volle bewust geworden. Daarom zijn in veel landen operaties gestart die tot doel hebben de erfenis uit het verleden weg te nemen of te beheersen.

Het inzicht omtrent de bodemonderzoeks- en bodemsaneringsproblematiek is tot stand gekomen door ontleding c.q. door het afbreken van complexe zaken in reeksen min of meer eenvoudige deelproblemen. Deze deelproblemen worden over het algemeen binnen één bepaalde discipline bestudeerd. Toch is vrijwel iedereen het er over eens dat bij het genereren van oplossingen meerdere disciplines tegelijkertijd dienen te worden betrokken. We hebben echter in zowel de publieke als in de private sector met een steeds verdergaande decentralisatie van verantwoordelijkheden te maken. Dit feit is eerder in het voordeel van een de klassieke monodisciplinaire aanpak waarin deelproblemen onafhankelijk van elkaar worden bestudeerd dan van een cross- of transdisciplinaire aanpak waarin juist de interactie tussen de relevante disciplines nadrukkelijk aan de orde komt. Hoewel de cross- of transdisciplinaire aanpak makkelijk met de mond beleden kan worden is het in de praktijk brengen ervan dus aanzienlijk moeilijker.

In de literatuur worden op diverse plaatsen handreikingen gegeven. De management-science-goeroe Michael Hammer wijst er op dat het voortborduren op bestaande gewoonten en werkwijzen zelden leidt tot significante efficiency- en effectiviteitverbeteringen en onderstreept het belang van de procesoriëntatie. In concreto: niet langer de subprocessen of deelproblemen maar het proces als geheel dient centraal te staan. De systeemliteratuur onderscheidt een drietal verschillende perspectieven van waaruit de effectiviteit van een organisatie kan worden beschouwd: het micro perspectief, het meso perspectief en het macro perspectief. Wanneer het micro perspectief als uitgangspunt wordt genomen, dan beschouwen we vooral de werkplek van een individu binnen de organisatie en verbeteringen betreffen derhalve altijd de prestaties van een individu. Het meso perspectief beschouwd het werkproces als geheel en spitst zich toe op de coördinatie tussen alle bij het proces betrokken individuen. Wanneer we de effectiviteit van de organisatie vanuit het macro perspectief beschouwen dan praten we over het gezamenlijke doel van meerdere organisaties. Hammer wijst dus sterk in de richting van het meso of macro perspectief.
Als basis voor de modellering van het bodemonderzoeks- en bodemsaneringsproces heb ik voor het besluitvormingsmodel van Mintzberg, Raisinghani en Théorêt gekozen. Het model onderscheidt een identificatiefase, een ontwikkelingsfase en een

selectiefase. Deze fasen kunnen overigens in verschillende volgorden worden doorlopen, waarbij het dus ook mogelijk is dat een fase wordt overgeslagen.

In dit proefschrift worden de verschillende fasen van het besluitvormingsproces doorlopen. In de hoofdstukken 3 en 4 wordt stilgestaan bij de identificatiefase. Binnen het bodemonderzoeks- en bodemsaneringsproces wordt deze fase over het algemeen met karakteriseringsfase aangeduid en bemonstering, chemische analyse en data-interpretatie vormen hier de karakteristieke elementen. In hoofdstuk 3 wordt aangegeven dat we ook *binnen* de identificatiefase of karakteriseringsfase te maken hebben met ontwikkelen en selecteren. De wijze van identificatie is namelijk afhankelijk van wat en waarom we wensen te identificeren. Om deze vragen te kunnen beantwoorden is een checklist opgesteld waarin wordt geïnformeerd naar het doel van de bemonstering, de te bemonsteren populatie, de te bepalen variabelen, de gewenste betrouwbaarheid, de te gebruiken bemonsterings- en analysemethoden en de gewenste bemonsteringsstrategie. Met de antwoorden kan door middel van *search* and *screen* worden gezocht naar bestaande en geschikte karakteriseringsoplossingen. Indien er meerdere oplossingen bestaan dient er voor één oplossing te worden gekozen. Om tot een keuze inzake de karakterisering te kunnen komen heb ik gebruik gemaakt van beslisbomen. Bij beslisbomen treft men vaak een aantal elkaar opvolgende beslismomenten aan. Dit houdt in dat het beslissingsprobleem uit een aantal stappen bestaat, waarbij na elke stap moet worden gewacht op het intreden van een bepaalde omstandigheid. Om te bepalen welke de verstandigste weg (keuze) is die moet worden ingeslagen, moet een overzicht gemaakt worden van de opbrengstbedragen samenhangend met de verschillende keuzemogelijkheden. De belangrijkste conclusie van hoofdstuk 3 is dat investeren in het karakteriseren wel degelijk rendement kan opleveren. De bewering dat nauwkeuriger karakteriseren altijd een beter rendement betekent gaat echter niet op.

In hoofdstuk 4 wordt de geostatistiek als belangrijk hulpmiddel bij de data-interpretatie geïntroduceerd. In de mijnbouw en petroleumwinning worden geostatistische methoden en technieken al enige tientallen jaren gebruikt voor voorraadschatting en karakterisering. Binnen het bodemonderzoeks- en bodemsaneringsproces zijn deze methoden eveneens geschikt om de verontreinigingssituatie in kaart te brengen. De verontreinigingssituatie vormt een eerste indicatie ten aanzien van de te volgen aanpak en de te verwachten kosten van de aanpak en is daarom essentieel voor de besluitvorming. Om deze redenen is het merkwaardig dat men dikwijls genoegen neemt met 'uit-de-losse-pols' getekende situatieschetsjes. Een nadeel van de geostatistische interpretatie is de relatief grote hoeveelheid benodigde data. Een voordeel van het gebruik van geostatistische methoden is het feit dat men ook inzicht in de onzekerheid van de schatting krijgt. Met andere woorden:

(financiële) risicocalculatie krijgt een fundament en is niet meer het resultaat van de een of andere 'natte-vinger-methode'.

In hoofdstuk 5 en 6 heb ik mij bezig gehouden met de ontwikkelingsfase van het bodemonderzoeks- en bodemsaneringsproces. Er zijn verschillende technieken om informatie te verkrijgen en te representeren. Om het uitwerken van ex situ en in situ bodemsaneringsalternatieven te kunnen beschrijven, is in eerste instantie een literatuurstudie gedaan. Hierbij komen vakspecifieke begrippen en definities en ook hun onderlinge relaties aan het licht. De literatuurstudie bleek een belangrijk hulpmiddel bij het bepalen van de kennis op domeinniveau. In tweede instantie zijn er interviews afgenomen met experts, met als doel die informatie te verkrijgen die niet was verkregen of niet kon worden verkregen uit de literatuur. Voor het representeren van de verkregen informatie en kennis is gebruik gemaakt van detail-processchema's. De eerste versies van deze schema's bleken een handig hulpmiddel bij de daaropvolgende interviews.
In grote lijnen kwam de informatie, verkregen uit de verschillende interviews, overeen. Het inzichtelijk maken van het werkproces en het formuleren van een uniforme werkwijze kon derhalve gemakkelijk worden gerealiseerd. Bij de realisatie van een kennissysteem kan worden gekozen voor een presenterend, een ondersteunend of een interpreterend systeem. Een presenterend systeem draagt kennis en informatie over aan de gebruiker. Bij keuzemomenten wordt er door de gebruiker zelf gekozen. In een ondersteunend systeem geeft het systeem een suggestie, maar de gebruiker kan – onderbouwd – hiervan afwijken. Een interpreterend systeem tenslotte, maakt zelf een keuze zonder dat de gebruiker deze keuze kan beïnvloeden.
Alvorens men overgaat tot het uitwerken van zowel ex situ als in situ sanerings-alternatieven dient men de beschikking te hebben over gegevens over bodem-opbouw, aanwezige verontreiniging(en), het doel van de sanering en aanwezigheid van bebouwing en leidingen. Er is in voor zowel ex situ als in situ saneren gekozen voor een presenterend papieren systeem. Bij ieder keuzemoment wordt een uitleg gegeven, die het verschil aangeeft tussen de twee of meer alternatieven. De tekst motiveert dus de keuze. Als de inzichten behorende bij een bepaald keuzemoment niet eenduidig zijn, wordt er een beroep gedaan op de inzichten van de gebruiker. Het resultaat van het kennissysteem bestaat uit de constatering dat ex situ of in situ sanering op die lokatie niet mogelijk is, of uit een aantal uit te werken concepten.
Er is voor het uitwerken van zowel ex situ als voor in situ saneringsalternatieven voldoende gedetailleerd gemodelleerd om een goede indruk te geven waar het bij het oplossen van het probleem om draait. Aan de andere kant is de mate van detaillering weer zo laag dat er geen pasklare oplossingen uit het model "rollen". Er moet dus nagedacht worden over de zin en onzin van de verzameling mogelijke oplossingen. Er wordt inmiddels in een ander verband geëxperimenteerd met een geautomatiseerd systeem met een zelfde detailniveau, maar met een ondersteunend karakter.

In hoofdstuk 7 wordt de selectiefase onder de loep genomen. De aanleiding voor het ontwikkelen van bodemsaneringsalternatieven vormt meestal het opheffen van gebruiksbelemmeringen van de bodem. Veelal komt dit neer op het reduceren van het risico voor omwonenden en het reduceren of voorkomen van de aantasting van lokale ecosystemen. Deze *baten* gaan gewoonlijk gepaard met *kosten*: niet alleen in termen van geld, maar ook in kosten voor het (niet-lokale) milieu. Zo is voor veel saneringsalternatieven inzet van energie nodig, hetgeen gepaard gaat met luchtverontreiniging. Echter, er bleek geen goede methodiek beschikbaar te zijn voor het objectief beoordelen van verschillende effecten van bodemsaneringsvarianten. Teneinde hierin te voorzien werd de RMK-methodiek ontwikkeld. Met behulp van dit model kunnen de Risicoreductie, de Milieuverdienste en de Kosten van saneringsalternatieven worden bepaald en kunnen de verschillende alternatieven met elkaar worden vergeleken.
De berekeningen ten aanzien van de risicoreductie concentreren zich op de gevolgen van de bodemsanering op de locatie. Bij risicoreductie is het perspectief 'het voorkomen van negatieve effecten van de verontreiniging en de saneringsmaatregelen op objecten (inclusief mensen) in de omgeving van de verontreiniging'. In de gebruikte definitie van risico komt dit perspectief terug: risico betreft alle vormen van blootstelling aan de beschouwde bodemverontreiniging, zowel ten gevolge van de verontreiniging zelf als ook ten gevolge van de gebruikte saneringsvariant, die kunnen leiden tot negatieve effecten op de gezondheid van gebruikers van de beïnvloede omgeving, op de gezondheid van personeel betrokken bij de uitvoering van de saneringsvariant, op het lokale ecosysteem in de door de verontreiniging beïnvloede omgeving en op nader te definiëren objecten, welke niet zijn uit te drukken in kosten. Zoals reeds eerder aangegeven is risicoreductie veelal de drijfveer voor saneringsoperaties. Dit betekent dat indien de risicoreductie bij een bepaalde saneringsvariant onvoldoende is, deze variant afvalt, ongeacht de prestaties voor milieuverdienste en kosten. Het perspectief van risicoreductie is 'lokaal'.
Milieuverdienste concentreert zich op potentiële invloeden op het milieu van de verontreiniging of van de saneringsoperatie. Het uitgangspunt is dat negatieve gevolgen voor het milieu zo klein mogelijk moeten zijn en dat de grondstofvoorraden zo veel mogelijk beschikbaar moeten zijn voor toekomstige generaties. Milieuverdienste beoordeelt de sanering vanuit het perspectief van het *algemeen belang*, dit in tegenstelling tot risicoreductie, dat zich richt op het lokaal belang. De definitie van milieuverdienste luidt: milieuverdienste is het resultaat van een geïntegreerde evaluatie van bovenlokale, niet-object-gerelateerde milieugevolgen veroorzaakt door een verontreinigde locatie en de sanering van deze locatie. De verdienste voor het milieu is het verschil tussen de aanvangssituatie en de eindsituatie van de sanering. Uitgangspunt van de evaluatie is het streven naar een duurzame samenleving. Milieuverdienste wordt daarom berekend op basis van

het beslag dat wordt gelegd op schaarse grondstoffen en ruimte, en de verwachte verandering van de milieukwaliteit. Naast de primaire effecten, zoals het elektriciteitsgebruik van pompen en het grondwatergebruik, worden ook secundaire effecten meegeteld. Dit zijn bijvoorbeeld emissies van zwaveldioxide door elektriciteitscentrales en het elektriciteitsgebruik van pompen in rioolwater-zuiveringsinstallaties indien afvalwater op het riool wordt geloosd. Tertiaire effecten, zoals energie nodig voor het maken of repareren van de pompen op de lokatie worden uit praktische overwegingen en omdat de verwachte effecten relatief klein zullen zijn buiten beschouwing gelaten.

Wellicht het meest herkenbaar is het kostenperspectief. Het onderdeel kosten binnen RMK bekijkt de sanering vooral vanuit het belang van de opdrachtgever van een saneringsoperatie. De opdrachtgever zoekt naar de kosteneffectiefste maatregel. Binnen de RMK-methodiek worden de kosten van saneringsvarianten zodanig berekend dat de kosten per jaar, vanaf de aanvang van de sanering, inzichtelijk worden gemaakt. Binnen de RMK-methodiek zijn kosten gedefinieerd als alle kosten die vanaf het beslismoment gedurende het vervolg van de sanering zullen worden gemaakt. Deze kosten omvatten de stichtingskosten, de doorlopende kosten, vervangingskosten, overhead en overige kosten.

De berekeningen binnen de RMK-methodiek leiden tot indices voor risicoreductie, milieuverdienste en kosten. Deze indices geven de prestatie van een saneringsalternatief op genoemde aspecten weer. Het is van belang dat effecten van bodemverontreiniging of -sanering niet zowel bij milieuverdienste als bij risicoreductie worden meegeteld. Dit soort 'dubbeltellingen' zal in theorie kunnen ontstaan als bij de verschillende perspectieven dezelfde doelen voorkomen. Bij het verwijderen van een verontreiniging telt voor risicoreductie *de afname van blootstelling bij objecten*, bij milieuverdienste telt *de toename van schone grond* en bij kosten tellen *de financiële consequenties van de verwijdering*. Hier is dus geen sprake van een dubbeltelling maar van een correlatie: elk thema kijkt vanuit zijn eigen perspectief naar de gevolgen van de handeling 'verwijderen van verontreiniging'

Het eenduidig zichtbaar maken van de prestaties van saneringsalternatieven op de verschillende aspecten (risicoreductie, milieuverdienste en kosten) leidt tot een betere belangenafweging van de verschillende actoren en, daardoor, tot een efficiënte besluitvorming. Uiteraard zijn niet álle specifieke belangen binnen de methodiek vertegenwoordigd. De eigenlijke beslissing blijft mensenwerk en RMK beoogt slechts deze beslissing te ondersteunen.

Het kiezen tussen verschillende saneringsalternatieven is een *meer-doelstellingsprobleem*. Zo zijn voor het ideale saneringsalternatief de verdienste voor het milieu en de risicoreductie maximaal, en de kosten minimaal. Het besluitvormingsproces bestaat vaak uit het zoeken naar consensus tussen de beslissers; een beslissing die een evenwicht zoekt tussen de doelstellingen op basis van de relatieve kwaliteiten van de alternatieven en de waarderingen van de bij de beslissing betrokken individuen (beslissers, belanghebbenden, publieke opinie). De RMK-methodiek genereert drie indexcijfers die het besluitvormingsproces kunnen ondersteunen. Een geïntegreerde RMK-index kan worden berekend door het gewogen gemiddelde van de berekende R-, M- en K-indices te bepalen. Elke beslisser zal hiervoor doorgaans voor elke bodemsaneringscasus zijn eigen set van gewichten vaststellen.

Concluderend kan worden gesteld dat het gebruik van het besluitvormingsmodel van Mintzberg, Raisinghani en Théorêt voor het bodemonderzoeks- en bodemsaneringsproces verhelderend is. Niet alleen worden de verschillende te doorlopen stappen beschreven en wordt aangegeven waar tijdens het proces interacties tussen de verschillende bijdragende disciplines moeten plaatsvinden, maar bovendien kunnen met behulp van het model betere keuzes ten aanzien van het toekomstige toegepaste onderzoek worden gemaakt.

Curriculum vitae

Joop Okx werd geboren op 2 april 1955 in Scheveningen. De middelbare school werd doorlopen aan de Europese School te Luxemburg, de Dalton-Scholengemeenschap te Den Haag en de Rijswijkse Openbare Scholengemeenschap te Rijswijk. Na een verblijf van zes jaar bij het 298 Squadron van de Groep Lichte Vliegtuigen werd in 1983 een aanvang gemaakt met de studie Fysische Geografie aan de Rijksuniversiteit Utrecht. In 1987 behaalde hij het doctoraal examen Fysisch Geografische Landschapskunde. Het afstudeeronderzoek bij prof.dr. P.A. Burrough betrof een geostatistisch onderzoek naar de ruimtelijke variatie in het zware metalengehalte in de bodems in de omgeving van de Zinkwit Nederland bv te Eijsden.

In 1989 trad hij, na twee jaar ervaring elders, als geostatisticus in dienst van Tauw te Deventer. In deze functie werd hij gesterkt in de overtuiging dat het bodemonderzoeks- en bodemsaneringsproces in vele opzichten als suboptimaal gekenschetst diende te worden. Op weg naar de "2.Jahrestagung des Deutschen Arbeitskreises für Mathematische Geologie und Geoinformatik" in Freiberg in de Freistaat Sachsen werd hem door Alfred Stein de suggestie gedaan om door middel van een promotie-onderzoek aan de optimalisatie van het bodemonderzoeks- en bodemsaneringsproces te werken.

Het in 1993 aan de Landbouwuniversiteit Wageningen gestarte promotie-onderzoek werd gecombineerd met het uitoefenen van werkzaamheden verbonden aan zijn respectievelijke functies projectleider GIS en Geostatistiek, interim vestigingsleider Tauw Umwelt Mannheim en afdelingshoofd Research & Development bij Tauw. Zijn huidige functie behelst het leidinggeven aan en het vormgeven van het R&D-proces binnen Tauw, alsmede het uitvoeren en begeleiden van complexe en omvangrijke R&D-projecten. Hiertoe wordt nauw samengewerkt met tal van binnen- en buitenlandse universteiten en instituten.

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