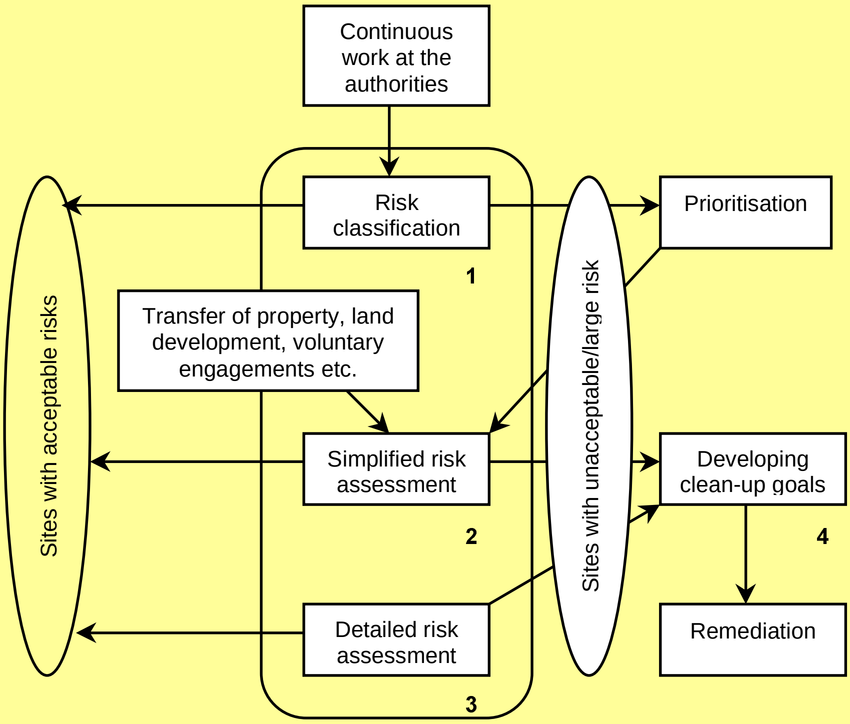
STATENS GEOTEKNISKA INSTITUT
SWEDISH GEOTECHNICAL INSTITUTE



Decision Analysis under Risk and Uncertainty at Contaminated Sites

– A literature review

JENNY NORRMAN

Varia 501

LINKÖPING 2001

Varia	Statens geotekniska institut (SGI) 581 93 Linköping
Beställning	SGI Litteraturtjänsten Tel: 013–20 18 04 Fax: 013–20 19 09 E-post: info@swedgeo.se Internet: www.swedgeo.se
ISSN ISRN	1100-6692 SGI-VARIA--01/501--SE
Projektnummer SGI Dnr SGI	10819 1-9912-717

Swedish Geotechnical Institute, SGI
Chalmers University of Technology, CTH

SGI Varia 501

Decision Analysis under Risk and Uncertainty at Contaminated Sites

A literature review

Jenny Norrman

FOREWORD

Contaminated land and groundwater is a problem of growing concern in our society. An increased environmental awareness, addressed in Sweden's new environmental legislation (*Miljöbalken*, 1999-01-01), has resulted in land and water contamination now being a major factor in land use planning and management, real estate assessment, and property selling. Investigation and remediation of contaminated areas are often associated with high costs. The Swedish EPA currently estimates that there are 22 000 contaminated sites in Sweden, of which approximately 4000 are in need for remediation.

The high costs and large number of contaminated sites are strong incentives for cost-efficient investigation and remediation strategies. *Miljöbalken* states that the environmental value of the remediation process must be higher than the investment costs. Critical issues to be addressed in order to meet the intentions of *Miljöbalken* and to provide cost-efficient handling of contaminated sites to landowners, operators, and the society are:

- What level of certainty is required to make sound decisions at a specific site, i.e. where, how and to what extent should sampling be performed?
- What remediation strategy is the most favourable with respect to investment costs and efficiency to meet specific clean-up goals?

Risk-based decision analysis is a theoretical approach to handle benefits, investment costs, and risk costs in a structured way to identify cost-efficient alternatives. Today, decisions regarding investigation strategies and remediation alternatives are in Sweden most often taken without completely or openly evaluating the cost-efficiency of alternatives.

The present report is one of two literature reviews prepared within the project *Risk-based decision analysis for investigations and remedial actions of contaminated land (Riskbaserad beslutsanalys för undersökningar och åtgärder vid mark- och grundvattenförorenade områden).* The project is sponsored by the Swedish Geotechnical Institute (SGI) and carried out in co-operation between SGI and the Department of Geology at Chalmers University. The main purpose of the project is to evaluate and describe risk-based methods for cost-effective investigation and remediation strategies with respect to Swedish conditions. The two reports are:

- *Sampling strategies and data worth analysis for contaminated land*, by Pär-Erik Back (Department of Geology, Publ. B 486 and SGI, report Varia 500).
- *Decision analysis under risk and uncertainty at contaminated sites*, by Jenny Norrman (Department of Geology, Publ. B 485 and SGI, report Varia 501).

The main purpose of the reports is to provide comprehensive descriptions of the state of the art of decision analysis and sampling strategies for contaminated land. The reports form an important basis for future work, not only in the specific project, but also in the wide topic of risk-based decision analysis.

Göteborg 2000-12-20

Lars Rosén, Ph.D
Supervisor

CONTENTS

Foreword

Abstract

1. **Introduction** 6
   1.1. Background 6
   1.2. Aim 6
   1.3. The structure of the report 6

2. **Risk analysis** 7
   2.1. Risk assessment 9
      2.1.1. Human health risk assessment 9
      2.1.2. Ecological risk assessment 9
      2.1.3. Models for risk assessment 10
   2.2. Risk management 11
   2.3. Environmental standards 12
   2.4. Legislative frameworks for risk assessment at contaminated sites 14
      2.4.1. Swedish Environmental Protection Agency 15

3. **Decision analysis** 18
   3.1. Decision theory 18
   3.2. Decision models 20
   3.3. Expected Utility 21
      3.3.1. Regret theory 22
      3.3.2. Prospect theory 22
   3.4. Maxiprobability 22
   3.5. Minimax 22
   3.6. Maximin and maximax 23
   3.7. Cost-benefit analysis 23
      3.7.1. Problem areas of CBA 24
      3.7.2. Valuation of non-market goods 25
   3.8. Decision criteria for uncertainty 29
   3.9. Social decision theory 29

4. **Uncertainties** 30
   4.1. Introduction 30
      4.1.1. Estimations of uncertainties 31
   4.2. Uncertainties associated with the geological system 32
      4.2.1. Examples of studies on parameter uncertainty in geological data 33
   4.3. Uncertainties in conceptual models 34
      4.3.1. Quantification of model uncertainty 36
   4.4. Uncertainties in risk assessment 36
      4.4.1. Examples of studies on uncertainty in risk assessment 37
      4.4.2. Uncertainty in environmental standards 38

5. **Decision-making frameworks applied at contaminated sites** 39
   5.1. Optimisation 39
      5.1.1. Early works on decision analysis frameworks with optimisation 40

5.1.2. Early works on data worth analysis 41
5.2. Early works on risk management frameworks 42
5.3. Risk-cost-benefit analysis 43
5.3.1. Examples of case studies with the application of risk-cost-benefit analysis 46
5.3.2. Including uncertainties in C_f 48
5.3.3. Including Regret 49
5.3.4. Alternative decision criteria 50
5.3.5. Data worth analysis using different investigation strategies 50
5.4. Social decision theory 52

6. **Discussion** 54

7. **References** 58

Appendix A - Risk assessment methods 67
References 68

Appendix B - International networks for contaminated land 69
Identified research needs 69
References 71

Appendix C - Suggested further reading 72

ABSTRACT

Contaminated land and groundwater is a problem of growing concern in our society. The number of sites that may cause adverse effects to the environment and to human health is high. The uncertainties associated with the spreading of contaminants and the effects that can be caused by these are large. Additionally, the costs of investigation and remediation are high. Thus, there is a need for developing methods to be able to deal with these problems cost-effectively. With the aim to provide a scientific basis for the purpose of developing a risk-based decision framework at contaminated sites, a literature review has been done. Decision theory under different degrees of knowledge is described. Risk, in decision theory, is defined as a state of complete probabilistic knowledge, whereas in practice, decisions at contaminated sites are characterised by only partial probabilistic knowledge i.e. uncertainty. Different types of uncertainty and probability estimations are presented. Also a review of studies conducted where decision analysis is applied at different problems is included. The final section of the report is a discussion, which comments on the most important aspects of using a risk-based decision analytical approach at contaminated sites. The discussion is held at two different levels; 1) the appropriateness of weighing costs against uncertainties and 2) practical and theoretical difficulties to provide a useful input to decision situations with incomplete knowledge.

Keywords: CBA, contaminated sites, data worth analysis, decision analysis, groundwater contamination, risk-cost-benefit analysis, risk analysis.

1. INTRODUCTION

1.1. Background

Contamination from old industrial sites, older landfills and other smaller sites with a potential to cause adverse effects on the surrounding environment and to human health is a problem many countries quite recently started dealing with. Much effort has been put into the development of risk assessment methods for the purpose of evaluating contaminated sites for prioritisation and setting environmental guidelines. Contaminated sites are characterised by large uncertainties both due to variability in the geological setting but also due to an often limited amount of data. All contaminated sites are characterised by specific geological, hydrogeological and geographical circumstances and need to be evaluated more or less site-specific to be able to make sound decisions. Since sampling and remediation of contaminated sites often is very costly there is a need for cost-effective prioritisation, remediation and sampling of the site. Not including uncertainties in the analyses may lead to bad decisions and cause unexpected costs, both as additional remediation costs and as environmental or health effects. Risk-cost-benefit analysis, being of most interest to this study, is a formalised way to incorporate economical considerations and uncertainties into the evaluation of remediation and sampling strategies at contaminated sites. The motivation for doing this is that a large number of uncertainties are present in several stages of an analysis and that these should be made visible and compared with costs.

1.2. Aim

The aim of this report is to picture an overview of the role of risk, uncertainties and decision analysis in connection to contaminated soil and groundwater to provide the scientific basis for an ongoing research project "Risk-based decision analysis at soil and groundwater contaminated sites". Since decision analysis applied at contaminated sites is an interdisciplinary field, aspects of economics, statistics, toxicology, and other sciences should and will be discussed and accounted for, although in some cases very briefly. The content of this report is mainly directed towards the foundations of decision analytical frameworks and how uncertainties are introduced into decision-making. It should be noted that this report is done from a hydrogeological point of view. This statement is not meant to belittle the importance of other sciences, it is merely to openly state a fact that will colour the content of this report.

1.3. The structure of the report

Chapter 2 gives a brief introduction to what is known as risk analysis, including risk assessment, both human health risk assessment and ecological risk assessment. It includes a subsection on environmental standards and legislative frameworks for risk assessment. Next, chapter 3 gives an overview of the theoretical foundations for decision analysis. Cost-benefit analysis is viewed as decision analysis with no uncertain outcomes and is thus included in this part. Chapter 4 is devoted to the issue of uncertainties. Uncertainties play a major role in the development of a decision framework, from model uncertainties to parameter uncertainties and how uncertainties can be estimated. In Chapter 5 the aim is, with the background of preceding chapters, to present different decision frameworks that have been published in the scientific literature. Data worth analysis is also conceptually described. Chapter 6 is a discussion with some conclusions for the continuation of the research project. Appendix A contains a table of risk assessment methods. In Appendix B, some international networks for the issue of contaminated land is listed. Identified research needs by two of these networks is included. A list of suggested further reading that is not referred to in the text is presented in Appendix C.

2. RISK ANALYSIS

Typically, risk is defined in different ways for specific purposes. It is not possible to claim that one way of defining risk is always more adequate than another, since this will depend on the purpose of defining risk. Instead it is important to understand the aim with defining risk in a certain way.

Risk is often referred to as the combined effect of the probability of a harmful event to occur and the magnitude of the consequence (Hartlén et al., 1998; Andersson & Lindvall, 1995; Hamilton, 1996;). Figure 2.1 gives a general visualisation of the concept of risk in the form of a risk matrix. Generally, a risk can only be postulated if: a) a hazard exists, b) a pathway occurs via which the effects of that hazard can be transmitted, and c) a target or receptor is exposed to doses of the contaminant hazard (Cairney, 1995). Thus, the casual chain of hazard → pathway → target/receptor has to remain unbroken. And as Asante-Duah (1998) writes, a key underlying principle of risk assessment is that some risks are tolerable.

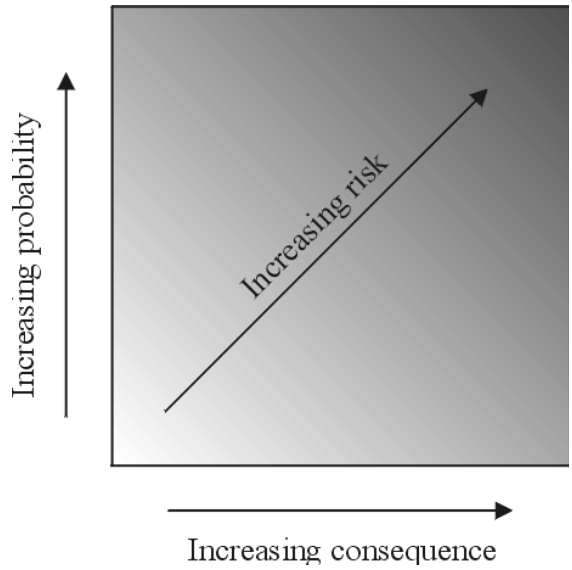


Figure 2.1. A general visualisation of risk in the form of a risk matrix.

The National Research Council (NRC, 1996) argue that risk should not be regarded as a quantifiable physical reality at all and gives a general definition of risk as being "A concept used to give meaning to things, forces, or circumstances that pose danger to people or to what they value. Descriptions of risk are typically stated in terms of likelihood of harm or loss from a hazard and usually include: an identification of what is "at risk" and may be harmed or lost (e.g., health of human beings or an ecosystem, personal property, quality of life, ability to carry on an economic activity); the hazard that may occasion this loss; and a judgement about the likelihood that harm will occur."

For the purpose of human health risk assessment or ecological risk assessment, the risk is commonly defined as a probability for a certain event. Referring to environmental literature, Davies (1996) defines risk as the *likelihood* that injury or damage is or can be caused by a substance, technology, or activity (see also Asante-Duah, 1998).

Risk analysis is commonly the most general term, encompassing all activities aimed at understanding, analysing, and managing risks. The National Research Council (NRC, 1996) identifies two fundamental phases in risk analysis namely risk assessment and risk management. Risk assessment is a set of analytical techniques for answering the question: How much damage or injury can be expected as a result of some event? A committee of the National Academy of Sciences (NAS, [1983][1]) devised a formulation of risk assessment as a four-step process; 1) hazard identification, 2) dose-response assessment, 3) exposure assessment, and 4) risk characterisation. Risk management on the other hand, considers the social, economic, and political factors involved in risk analysis (Davies, 1996; NRC, 1996; Massmann and Freeze, 1987a, b). This determines the acceptability of damage and what, if any, action should be taken. Figure 2.2 gives a traditional schematic overview of the two phases of risk analysis and its interaction with research.

NRC (1996) refers to risk assessment as being a process for understanding and risk management a process for action. Hartlén et al (1999) identifies an additional step following after the risk assessment in some frameworks, namely risk evaluation. Risk evaluation involves comparing and judging the significance of risk (Hartlén et al., 1999) and is often viewed as being a part of the risk management process. Davies (1996) adds two more concepts as being part of risk analysis, risk communication and comparative risk analysis (risk ranking) - CRA. Risk communication is conveying information about risk and CRA is a system for comparing different risks.

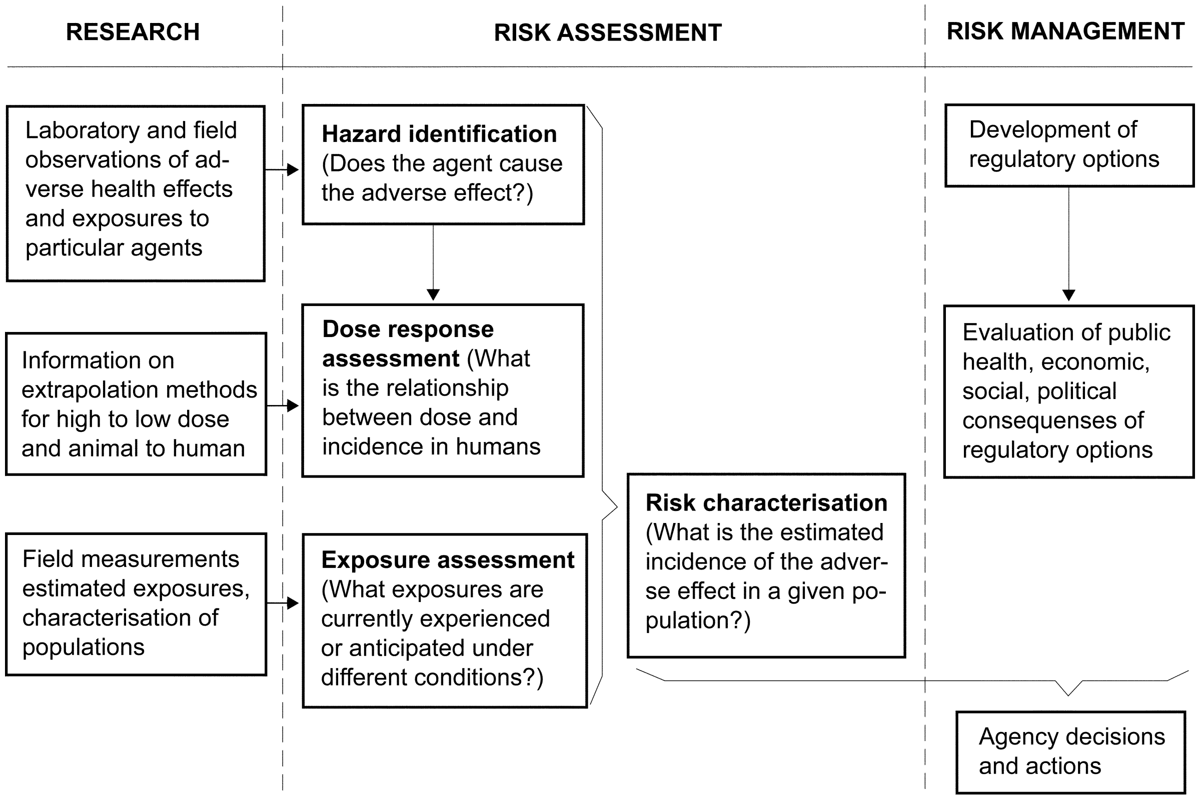


Figure 2.2. A schematic representation of the traditional view of risk analysis consisting of risk assessment with its four steps and risk management. After NRC (1996).

[1]Called “the Red Book” written the National Research Council of the Academy of Sciences in 1983 (Felter et al., 1998; Asante-Duah, 1998; Covello and Merkhofer, 1993; NRC, 1996; Davies, 1996).

2.1. Risk assessment

It is generally acknowledged that toxic material present or introduced in air, soil, water, food etc. that poses risk to humans and ecological health forms a great challenge to environmental management and prioritisation for action. Environmental risk assessment is developed as a formalised tool for scientists to evaluate the toxicity data for chemicals to which humans are, or may be, exposed and to attempt to identify and quantify potential risks to health (Felter et al., 1998). Risk assessment can help identify existing hazardous situations or problems, anticipate potential problems, provide a basis for regulatory control and corrective actions and to help gage the effectiveness of corrective measures or remedial actions (Asante-Duah, 1998). The use of risk assessment in setting environmental standards is discussed in part 2.3. Section 2.4 describes how risk assessment is recommended by the Swedish Environmental Protection Agency (Swedish EPA) at contaminated sites.

2.1.1. Human health risk assessment

The pathways for exposure of toxic substances to humans are generally categorised as: inhalation of fugitive dust and volatile compounds, ingestion of water, soil, crops and diary and beef products, ingestion of soil and sediment by accident or by pica behaviour[2], and dermal contact to soil and water (Asante-Duah, 1998). Equations to compute the exposure of each exposure route can be found in literature (e.g. Asante-Duah, 1998). The effects are measured in toxicity parameters for non-carcinogenic and carcinogenic effects. The cancer risk is usually expressed as an excess lifetime probability to develop cancer over a population. In Europe, the theoretical tolerable excess lifetime cancer risk typically used in the context of genotoxic carcinogens on contaminated sites ranges from 10^{-6} to 10^{-4} per substance (Ferguson et al., 1998).

The non-cancer risk is usually expressed by the hazard quotient (HQ) and/or the hazard index (HI). The HQ is defined as the ratio of the estimated chemical exposure level to the route-specific reference dose. The HI is used for the aggregate non-cancer risk for all exposure pathways and all contaminants associated with a potential environmental contamination problem. If the HI exceeds unity (1) there may be potential for adverse health effects (Asante-Duah, 1998).

2.1.2. Ecological risk assessment

The ecological risk assessment evaluates the probability or likelihood that adverse ecological effects will occur (or have occurred or are occurring) as a result of exposure to stressors from various human activities (Smrchek and Zeeman, 1998). The authors define the term stressor as a description of something chemical, physical or biological in nature, which can cause adverse effects on non-human ecological components ranging from organisms, populations and communities, to ecosystems. The ecological risk assessment process is slightly modified from that of human health risk assessment (Smrchek and Zeeman, 1998; Asante-Duah, 1998).

There are in principle two approaches, top-down and bottom-up. Although the top-down approach, which evaluate toxic effects in a ecosystem perspective is preferred it is difficult to carry out. In the bottom-up approach, hazards and risks identified are extrapolated from laboratory tests in organisms to populations, communities and even ecosystems. Ecotoxicology tries to combine the two approaches (Smrchek and Zeeman, 1998; Asante-Duah, 1998).

---

[2] Pica behaviour (eating of non-food items, including soil) is common in many toddlers and children (Canadian Association of Physicians for the Environment, 2000).

2.1.3. Models for risk assessment

The human health risk assessment model most referred to is that by the National Research Council of the Academy of Sciences in 1983 (NAS-NRC), also called the red book (Felter et al., 1998; Asante-Duah, 1998; Covello and Merkhofer, 1993; NRC, 1996; Davies, 1996). The model has been widely used by government agencies in the USA for assessing the risks of cancer and other health risks that result from exposure to chemicals. Four steps are proposed in a complete risk assessment, hazard identification, dose-response assessment, exposure assessment, and risk characterisation.

Hazard identification

Hazard identification involves a qualitative assessment of the presence of, and the degree of hazard that an agent could have on potential receptors. It involves an evaluation of the appropriateness, nature, quality and relevance of scientific data on the specific chemical; the characteristics and relevance of the experimental routes of exposure; and the nature and significance to human health of the effects observed (Felter et al., 1998). A quantification of the concentration at which they are present in the environment should also be conducted.

Hazard identification of non-cancer end-points and carcinogens differs slightly. Hazard identification for non-cancer end-points depends much on professional judgement whether to judge an response as adverse or not since toxic chemicals often elicit more than one adverse effect. To determine whether a compound has the potential to elicit a carcinogenic response or not many types of information can be used: epidemiological information, chronic animal bioassays, etc. Classification schemes for carcinogenicity has been developed for this purpose (Felter et al., 1998).

Dose-response assessment

A dose-response assessment is a further evaluation with specific emphasis on the quantitative relation between the dose and the toxic response. Information for doing this assessment can be derived from different studies on human exposures, epidemiology etc. Most important is that the dose-response assessment is based on data of sufficient quality, as judged by experts.

Exposure assessment

The exposure assessment estimates the magnitude of actual and/or potential receptor exposures to environmental contaminants, the frequency and duration of these exposures, the nature and size of the populations potentially at risk, and the pathways by which the risk group may be exposed. To complete a typical exposure analysis for an environmental contamination problem, populations at risk are identified, and concentrations of the chemicals of concern are determined in each medium (air, water, soil etc.) to which potential receptors may be exposed (Asante-Duah, 1998).

Risk characterisation

Risk characterisation involves integration of information from the first three steps to develop a qualitative or quantitative estimate of the likelihood that any of the hazards associated with the agent of concern will be realised in exposed people (Felter et al., 1998). It should also include an elaboration of uncertainties associated with the risk estimates. It includes a discussion of the assumptions made and the overall quality of data. It is here where the risk assessment results are expressed.

Other models for risk assessment
An example of another model for risk assessment was presented by Covello and Merkhofer (1993). Their model of risk assessment consists of; release assessment, exposure assessment, consequence assessment, and risk estimation. The risk chain is identified as; risk source release processes – exposure processes – consequence processes. They note that their model is similar to the NAS-NRC model but list some significant differences of which the most important is that the NAS-NRC model includes hazard identification in risk assessment when the above mentioned authors choose to view it as a separate process that is necessarily conducted prior to risk assessment.

A complete risk assessment requires the application of a large and diverse set of methods as noted by Covello and Merkhofer (1993). In Appendix A is a list of a number of methods categorised according to their proposed risk assessment model. It is obvious, whichever models for risk assessment and methods for the different steps of such an assessment one chooses, that a number of skills and expert knowledge are required. This makes risk assessment difficult since it requires communication between different experts and the public. In turn, rightly done it provides a good basis for discussion and communication.

2.2. Risk management

A guide developed by the American Chemical Society & Resources For the Future (1998) points out three tasks as essential in risk management.

1. To determine what hazards present more danger than society is willing to accept.
2. To consider what control options are available.
3. To decide on appropriate actions to reduce (or eliminate) unacceptable risks.

The risk assessment procedure as described in previous section is an input to risk management but fails to alone provide answers on how to make trade-offs between risks and costs or how to prioritise risks. Of course, risk management enters the political arena since the answer to such issues depends on society's values and priorities. Hansson (1989) concludes that risk decisions are part of the general political process and that expert assessments should be presented in a way that reflects the complexity of the subject matter, instead of repressing it. Decision theory is introduced in chapter 3. Decision theory has developed as a tool to aid in complex situations where decisions must be made.

Subsections of risk management are risk perception and risk communication. There is plentiful literature in these areas, many of which the contributors are psychologists and philosophers by profession.

Risk perception
Hansson (1989) presents eight factors of risk comparisons as different dimensions of risk. Those are 1) the character of the negative consequences, 2) control and free choice, 3) individual and collective perspective on risks, 4) large disasters and probability, 5) the time factor, 6) decisions under uncertainty, 7) new and old risks, and 8) the availability of knowledge. These are all factors that influence our understanding and perception of risk.

Risk communication
Risk communication is directed at increasing the public's knowledge of risk issues and participation in risk management (American Chemical Society, 1998). Examples are warning

labels, public databases, and public hearings on these issues. It is a dialogue between the interested parties – risk experts, policy makers, and affected groups of the public.

2.3. Environmental standards

Environmental standards can, according to Whitehouse & Cartwright (1998), be used for the following purposes:

- Environmental benchmarks against which environmental monitoring data can be assessed;
- Setting goals for pollution control activities;
- Acting as triggers for remedial action;
- Environmental management tools which can be applied across different locations and times.

Whitehouse and Cartwright (1998) uses a simplified model from Barnett and O'Hagan [1997] to show where different types of regulations are used (figure 2.3). The pollutant is generated (actions), received by the environment (exposure), biota and/or humans become exposed to the pollutant (contact) and finally, effects show (effects). Action-based standards can be exemplified by restrictions on the use of certain products, standards applied to the point of entry by fixed uniform emission limits in air, water and soil. Standards which apply to the point of contact does not depend on how the contamination entered the medium, merely on critical end-of-pipe concentrations after a defined "mixing zone".

Effect-based standards are rarely used and are expressed in terms of toxicity rather than chemical concentrations, taking into account simultaneous exposure to a range of contaminants (Whitehouse & Cartwright, 1998). According to Johannesson (1998) limit values and guide values for exposure, emission, concentration etc. are important tools for authorities to reduce risks.

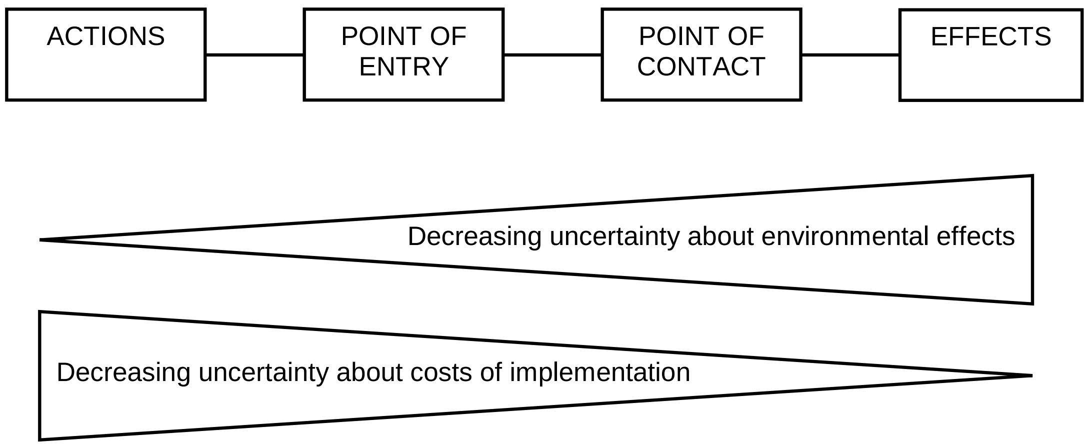


Figure 2.3. Positions at which environmental standards may operate (After Whitehouse and Cartwright, 1998).

For the purpose of this literature study, standards that apply to the point of exposure is of most interest. Whitehouse and Cartwright (1998) present three models to obtain soil standards: the mass-balance approach, the technical approach and by "risk-based" models. The mass-balance approach applies to agricultural soils and aims at maintaining the status quo between inputs to the soil and outputs in crops. The technical approach uses human toxicity and eco-toxicity effects data to determine soil standards. Finally, "risk-based" models are similar to the technical approach but includes physico-chemical models for the transfer of pollutants to the identified receptors, thus a risk assessment approach. King (1998) notes that risk assessment is increasingly being used in the UK and in the EU as an aid in standard setting.

According to Wentsel (1998), the US EPA implements narrative directives primarily through three different approaches: health-based standards, technology-based standards and no unreasonable (balanced) risk standards. Health-based standards regulate the protection of human health or the environment without regard to technology or cost factors. It is a zero risk standard. Technology-based standards require best practicable control technology, best available technology for pollution reduction or treatment and often weighs the effectiveness of a technology against costs, rather than risk reduction. Balanced risk-based standards requires the balancing of risk assessment results against cost-benefit to determine the risk management approach.

Wentsel (1998) and Whitehouse and Cartwright (1998) thus slightly differs in their definition on the different methods to derive standards. What is remarkable is that the US EPA explicitly requires weighting of costs and benefits.

Generic guidelines at contaminated sites in Sweden

The Swedish EPA has developed generic guideline values for contaminated soils in Sweden according to a health-risk based model (Statens Naturvårdsverk, 1996a, b). Generic guideline values are developed for three different kinds of land-use: sensitive use, e.g. land used for residential areas, kindergarten, agriculture, groundwater extraction, etc.; land with less sensitive use and groundwater extraction, e.g. land used for offices, industry, roads, etc.; less sensitive use as above but with no groundwater extraction.

For human health effects the following exposure pathways have been considered: direct intake of contaminated soil; dermal contact with contaminated soil and dust; inhalation of dust from the contaminated site; inhalation of vapours; intake of contaminated drinking water for land-use with groundwater extraction; intake of vegetables grown on the contaminated site (land for sensitive use); intake of fish from nearby surface water (land for sensitive use). Eco-toxicological effects both on the site and due to transport is taken into account. The basic principle of the Swedish EPA has been to choose the lowest value of human toxicological and eco-toxicological value. (Statens Naturvårdsverk, 1996b) Other countries also have risk-based guidelines. The Swedish EPA (Statens Naturvårdsverk, 1996a) mention the Netherlands, USA, Canada, UK, Germany and Denmark. The methods used are similar but the models are adapted to country-specific conditions.

Generic versus site-specific guideline values

Generic guideline values may in some cases be so low that it will be extremely costly or impossible to clean everything to such low concentrations, at least with existing remediation technologies. In recent literature there are several suggestions on how to model site-specific conditions to obtain more relevant site-specific limit values.

Nikolaidis et al (1999) suggests a methodology for site-specific mobility-based cleanup standards for heavy metals in glaciated soils. The methodology includes a detailed soil characterisation, laboratory mobility studies, and mathematical modelling and would be an alternative to the traditional, risk-based approach for establishing site specific cleanup criteria. In principal it is based on an assumption that not the whole amount of contaminant in the soil will be mobilised, but adsorbed to some extent and also naturally attenuated. The study was done with Chromium and modelled with surface complexation reactions.

Hetrick and Pandey (1999) developed and used a vadose-zone soil-leaching mathematical model to help define deep-soil preliminary remediation goals (PRG) at the Portsmouth Gaseous Diffusion Plant in Portsmouth, Ohio. Deep soil was defined as unsaturated soil from below the ground surface to depths exceeding 10 ft. (approximately 3.3 m). The study defined those deep-soil concentrations that would most likely not cause groundwater contamination in excess of US EPA guidelines. They conducted deterministic studies and complemented with probabilistic studies so that a higher degree of certainty could be placed on the predicted PRGs.

Mohamed and Côté (1999) used a decision analysis based model (DAPS 1.0, Decision analysis of polluted sites) to answer the question whether a polluted site should be remediated or not. Pathways are simulated via transport models and concepts of fuzzy set theory was adopted to account for uncertainty in the input parameter. They evaluate the carcinogenic risk (CR) and the hazard index (HI) and combine these to achieve a risk factor (RF). The risk factor is evaluated to determine whether action is necessary or not.

Al-Yousfi et al (2000) proposed a risk-based zoning strategy for soil remediation at an industrial site. High risk and medium risk areas were identified through direct human exposure scenarios (inhalation, ingestion, and dermal contact) and resources were allocated to these areas. The authors conclude that as opposed to simple comparison with soil cleanup standards, their strategy has facilitated the tasks of sorting out information and directing attention to zones of significant nuisance contribution.

2.4. Legislative frameworks for risk assessment at contaminated sites

When contaminated land is to be reclaimed there are, according to Cairney (1995), a range of interest groups who inevitably become involved:

- Developers, whose interest are in minimal costs and fastest development completions.
- Financial bankers, who wish to maximise their profits without undue exposure to any risks.
- Consultants, acting either for developers or for control bodies.
- Planning authorities and control bodies (acting for local authority and environmental protection interests) whose aims are to minimise adverse impacts and avoid the public being put at risk.
- The interested public, usually represented by local environmental pressure groups.

Obviously these groups have different perceptions and concerns, hence the risks that can affect the different groups will also differ.

Regulatory agencies can be seen as institutions with the aim to make the private decision-makers perspective more alike the societal decision-makers perspective. In economical terms, one may say that it has a distributional role. Regulatory agencies that are to put up laws and regulations have two main parties to consider; the public’s health and security and to provide a good climate for companies to operate within (Cairney, 1995).

2.4.1. Swedish Environmental Protection Agency

The Swedish Environmental Protection Agency (EPA) has estimated approximately 22,000 contaminated sites[3] in Sweden of which 12,000 are identified. Many of these sites are in need of remediation and the new law in 1999, the Environmental Code (*Miljöbalken*) is aimed at facilitating this task. The law includes (Norman, 1999):

1. An obligation to report and make public any detected contaminants of land and water.
2. The possibility to register property and impose restrictions on land use.
3. Rules on liability for the investigation and remediation of contaminated land. The liability rests in the first place with the person who caused the pollution, and then with the person who owns the contaminated land.

There are two main principles underlying the law. The first is the precautionary principle, meaning that actions to protect the environment may be needed before one is fully able to predict the consequences. It also means that risk assessments should involve a generous safety margin (Norman, 1999). The second key principle is the polluter pays principle. The Swedish EPA will provide more than 500 million SEK of governmental resources to the Swedish county authorities during a three-year period, starting 2000, for investigation and remediation of contaminated areas where no responsible part can be found. Still, the major part of necessary investments for investigations and remediation will come from responsible landowners and operators.

The risk assessment as recommended by the Swedish EPA is made without concern for cost or technology and involves identifying and describing the risk of adverse effects on human health or the environment from a given site. The reason for making risk assessments are, according to Norman (1999); to make it possible to rank the contaminated sites in order to prioritise action, to assess the state and seriousness of the situation today and in the future, and to assess contamination levels that may remain at the site. Figure 2.4 gives the outline of the risk assessment scheme in Sweden.

Figure 2.4 illustrates three types of risk assessment procedures (Norman, 1999):

1. Risk classification, made in connection with inventories of contaminated sites.
2. Simplified risk assessment based on generic and/or sector-specific guideline values for determining risks.
3. Detailed risk assessment based on site-specific values.

Future work identified is to concentrate on methods on how to develop clean-up goals, principles and methods for detailed risk assessment, development of generic guidelines for groundwater and sediments, and to provide support to county administrative boards and municipalities with their work on remediation of contaminated sites (Norman, 1999).

[3] A contaminated site is defined as "…a landfill site or area of soil, groundwater or sediment which is so contaminated by a point source that concentrations substantially exceed local/regional background levels."

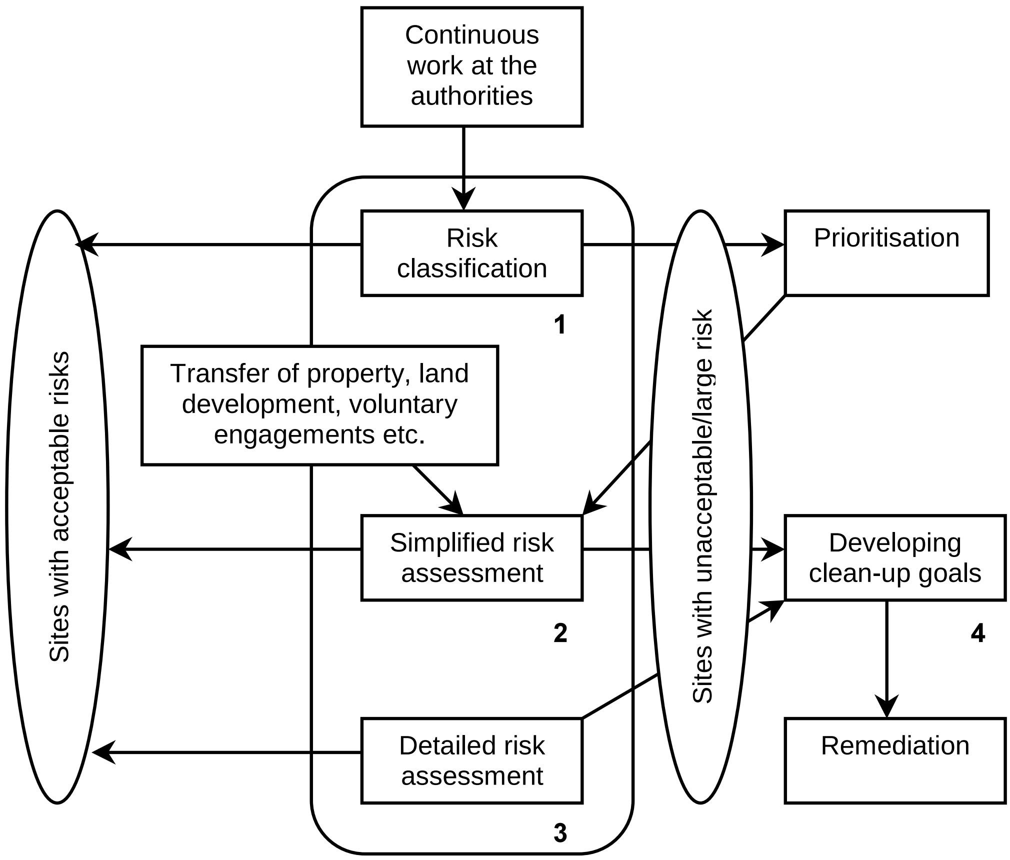


Figure 2.4. Risk assessment of a contaminated site as recommended by the Swedish EPA. (After Norman, 1999).

The Swedish EPA has developed a method of surveying contaminated sites ("MSCS" - Method of Surveying Contaminated Sites[4]). The following gives a brief description of the factors that are investigated in the risk assessment of contaminated sites as recommended by the Swedish EPA (Statens Naturvårdsverk, 1999b). The different steps are thoroughly described regarding methods and analysis. Forms produced by the Swedish EPA should be filled out during the assessment procedure as a way to document investigations following a standard. The main objective is that identification and assessment of contaminated sites will be qualified and in unity, with the aim to prepare for further investigations and eventually remedial actions. Figure 2.5 shows the risk classification matrix to be used in the assessment. The measure of risk is here qualitative.

Risk class 1. “Very high risk”
Risk class 2. “High risk”
Risk class 3. “Moderate risk”
Risk class 4. “Low risk”

The y-axis is represented by spreading conditions from constructions, to constructions, in soil and groundwater, to surface water, in surface water, and in sediments. Each transport pathway is judged as being low, moderate, high, or very high. The division into these categories is done by the help of principles, forms and tables.

[4] *In Swedish:* MIFO – Metodik för Inventering av Förorenade Områden.

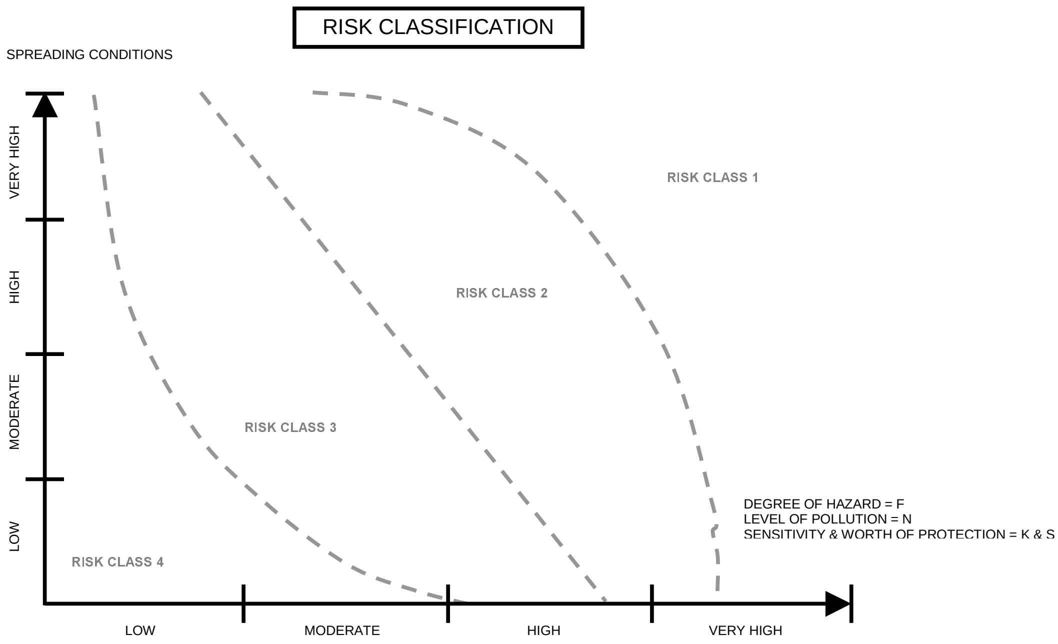


Figure 2.5. Risk classification matrix according to the Swedish EPA. (After Statens Naturvårdsverk, 1999)

The x-axis is a combination of four different factors: the degree of hazard posed by the pollutants (F), the level of pollution (N), sensitivity (K) and the degree to which the area is worthy of protection (S) (abbreviations in Swedish).

The degree of hazard (F) is assessed from a number of hazard classes, put up by the National Chemicals Inspectorate. Necessary to know is which substances are present at the site or which substances can be expected to be present at the site. If a substance is not presently classified, further investigations should be done. The level of pollution (N) is assessed from how much the generic guidelines are exceeded. Generic guideline values exist for 36 contaminants or groups of contaminants in soil. Sector-specific guideline values related to petrol stations exist for 18 contaminants or groups in soil and 12 groundwater contaminants (Norman, 1999).

Sensitivity (K) is the degree to which humans and the environment will be exposed to the contaminants today and in the future. Low sensitivity is where humans do not get exposed, e. g. a fenced-in site with no ongoing activity. Highly sensitive, on the other hand are areas where humans live permanently, children are exposed or where groundwater or surface water is used for drinking water purposes. Worth of protection (S) is a categorisation from the principles that already polluted areas have a low value whereas e.g. national parks, areas with specific species or eco-systems present have a high value.

The methodology can be used for all sorts of contaminated sites (old industrial sites, landfills etc.) with varying amount of data and varying quality of available data. When large uncertainties exist the classification should be stricter, that is, be on the "safe side". The (subjective) judgements should be based on a "probable but bad case" and not on "worst case" as this may overestimate the risk gravely.

3. DECISION ANALYSIS

Keeney (1982, 1984) thinks of decision analysis as "a formalization of common sense for decision problems which are too complex for informal common sense." Dakins et al. (1994) describes it as "Decision analysis is a technique to help organize and structure the decision maker's thought process, elicit judgments from the decision maker or other experts, check for internal inconsistencies in the judgements, assist in bringing these judgements together into a coherent whole, and process the information and identify a best strategy for action". A more technical definition is "a philosophy, articulated by a specific set of logical axioms, and a methodology and collection of systematic procedure, based upon those axioms, for responsibly analysing the complexities inherent in decision problems" (Keeney, 1984). Risk management decisions are often complex decision situations with several affected groups and different types of information.

3.1. Decision theory

Decision theory is according to Hansson (1990) one of the truly interdisciplinary sciences. The major sciences involved in decision theory was listed by Patrick Suppes in 1961. Table 3.1 is a modernisation of the list done by Hansson (1990). The list is divided into four groups depending on normative or descriptive emphasis or whether individual or group decisions are of interest. A normative decision theory is a theory about how decisions should be made, whereas a descriptive theory is a theory about how decisions are actually made. Individual decision-making may as well apply to decision-making by groups, given that the group acts as if it were a single individual. A collective decision theory, on the other hand, is a theory that models situations in which decisions are taken by two or more persons, who may have conflicting goals or conflicting views on how the goals should be achieved. Most studies in collective decision theory concerns voting or bargaining or similar. Hansson (1990) places risk analysis in all four squares in Suppes' diagram.

Table 3.1. Hanssons (1990) modernised version of Suppes' table [1961] of the major sciences involved in decision theory.

	Individual decisions	**Group decisions**
Normative theory	Classical economics Statistical decision theory Moral philosophy Jurisprudence Game theory Artificial intelligence Optimisation Fuzzy set theory Risk analysis	Game theory Welfare economics Political theory Social decision theory Risk analysis
Descriptive theory	Experimental decision studies Learning theory Survey studies of voting behaviour Artificial intelligence Risk analysis	Social psychology Political science Risk analysis

Another categorisation in decision theory is the degree of knowledge under which a decision is made. Table 3.2 gives a scale of knowledge situations in decision problems. Hansson (1990) concludes that decision how to store nuclear waste is clearly an example of decision-making under uncertainty.

Table 3.2. Degrees of knowledge (Hansson, 1990).

Certainty	Deterministic knowledge
Risk	Complete probabilistic knowledge
Uncertainty	Partial probabilistic knowledge
Ignorance	No probabilistic knowledge

Hansson (1990) identifies three stages in the decision process. These are identification of the problem, development to define and clarify the options, and selection of alternative. Figure 3.1 shows the relationship between these stages and their routines, which is circular rather than linear.

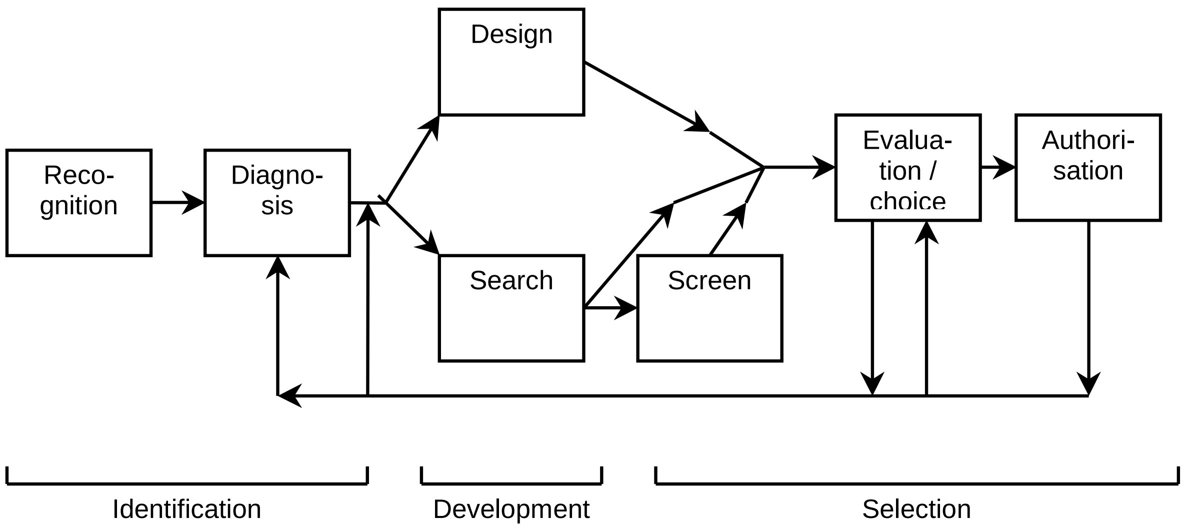


Figure 3.1. The decision process. Relationships between the phases and routines (after Mintzberg et al. [1976] in Hansson, 1990).

Keeney (1982) takes a more narrow approach, while discussing the methodology of decision analysis. He identifies four steps: 1) structure the decision problem, 2) assess the possible impacts of each alternative, 3) determine preferences of decision makers, and 4) evaluate and compare alternatives (see figure 3.2).

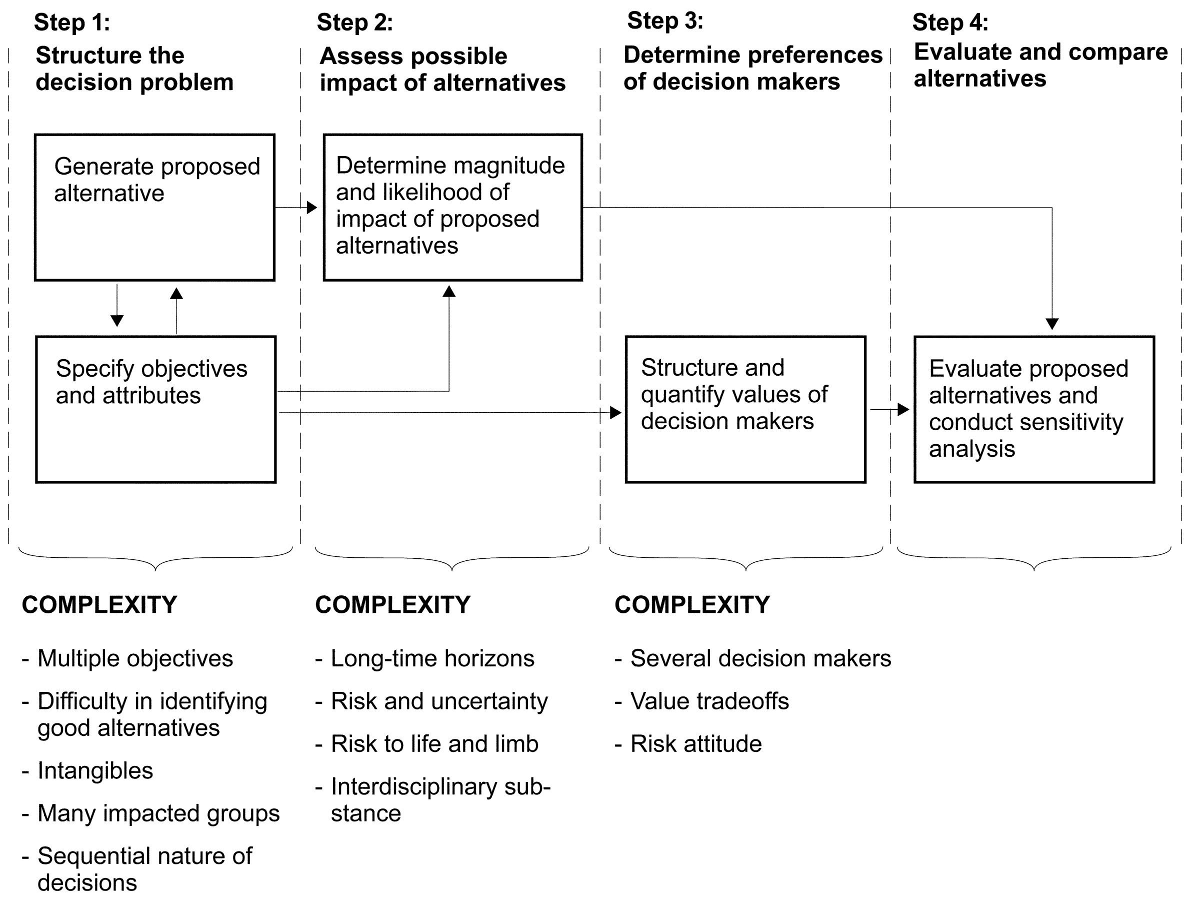


Figure 3.2. Schematic representation of the steps of decision analysis. Some features of complexity is shown in the different steps (from Keeney, 1982).

Hansson (1990) notes that the evaluation-choice routine in the selection stage (see figure 3.1) is treated by far most in decision theory. The neglect of the other parts of decision theory may be because they are not as readily accessible to mathematical treatment as the evaluation-choice routine. Hansson (1990) argues that one of the most urgent tasks for normative decision theory is to extend its interest to the other routines of a decision process. Important though, is to point out that decision theory does not enter the scene until the ethical or political norms are already fixed. Thus, decision theory provides methods for an environmental agency to minimise toxic exposure, but the basic question whether it should try to do this is not treated.

A decision is unstable if the very fact that it has been made provides a sufficient reason to change it. Hansson (1990) writes that there is a debate whether a rational decision must be stable. There are also suggestions on how to incorporate stability tests in decision analysis.

3.2. Decision models

Depending on what degree of knowledge is present, different decision models can be applied. However, the necessary information for a specific decision model is not always available. Thus in practice, decision-makers are often forced to make decisions under uncertainty and under pressure against time (Johannesson, 1998). Part 3.3 to 3.9 describes different decision models. Johannesson (1998) lists different kind of knowledge that are present in the decision models he presents, see table 3.3.

Table 3.3. Different kind of knowledge listed by Johannesson (1998).

1a	The decision alternatives are known, and so is the outcome of each alternative.
1b	The decision alternatives are known, and so is the probability of different outcomes when choosing each of them.
1c	The decision alternatives and their possible outcomes are known.
2a	The utility[5] of different outcomes are known.
2b	It is possible to rank different outcomes according to their utility.

3.3. Expected Utility

The dominating model of decision-making under risk is the expected utility (EU) model. Criteria 1b and 2a in table 3.3 must be fulfilled. Every possible outcome is assigned a utility. The utility is weighted with the probability of the outcome and the objective of the decision model is to maximise the expected utility (MEU). Johannesson (1998) points out that the EU rule can be a too risk-taking strategy in situations which are unique or rare, or situations which include a possibility of catastrophic outcome.

Both the utilities and the probabilities can be either objective or subjective (Hansson, 1990). Expected utility theory with the use of both subjective utilities and subjective probabilities is commonly called Bayesian decision theory, or Bayesianism. *Bayesian decision theory is always under risk or certainty.* An operative limitation of Bayesianism is that no intersubjectively valid expected utilities can be calculated (Hansson, 1990). That is, if utilities are subjective they may (or will inevitably) differ from person to person.

On the other hand, objectivist expected utility has a weak normative standing since there are situations in which decision makers violate consistency, so called paradoxes. One of the most well known paradoxes is the one of Allais [1953] (Bell, 1982; Hansson, 1990). He showed that although most people select a prize of [A: $1 million for sure] rather than a gamble giving [B: a 10% chance at $5 million and a 90% chance at nothing], a majority of people prefer a gamble offering [D: a 10& chance at $5 million and a 90% chance at nothing] rather than a gamble of [C: an 11% chance at $1 million and an 89% chance at nothing]. This can be presented in decision matrixes, see figure 3.3.

	S_1 [0.10]	**S_2 [0.89]**	**S_3 [0.01]**
A	1 000 000	1 000 000	1 000 000
B	5 000 000	1 000 000	0

	S_1 [0.10]	**S_2 [0.89]**	**S_3 [0.01]**
C	1 000 000	0	1 000 000
D	5 000 000	0	0

Figure 3.3. Allais' paradox presented in two decision matrixes (after Hansson, 1990).

This behaviour is incompatible with being consistent with maximising the expected utility (in this case both subjective and objective). There are some broadened models of expected utility such as regret theory and prospect theory to deal with such cases.

[5] The utility of an outcome is a concept meaning the satisfaction, happiness or wellbeing of an outcome. The quantification of utilities is mostly done in monetary terms, although this may fail to reflect the true utility.

3.3.1. Regret theory

Bell (1982) suggests to incorporate regret in expected utility analysis because the single-attribute utility function seems to produce paradoxes, thus failing to describe some simple comparisons. Regret theory makes use of a two-attribute utility function, 1) utility of outcomes and 2) quantity of regret (Bell, 1982; Hansson, 1990). Hansson (1990) describes regret as "the painful sensation of recognising that 'what is' compares unfavourably with 'what might have been'." See further part 5.3.3.

3.3.2. Prospect theory

Hansson (1990) mentions prospect theory as another alternative to expected utility theory. It is especially developed for problems with monetary outcomes and objective probabilities. The monetary outcomes are replaced by a values function v, depending both of the (subjective) utility of the outcome and the present state of wealth (see figure 3.4). The objective probabilities are transformed to decision weights by means of a function π, between 0 and 1.

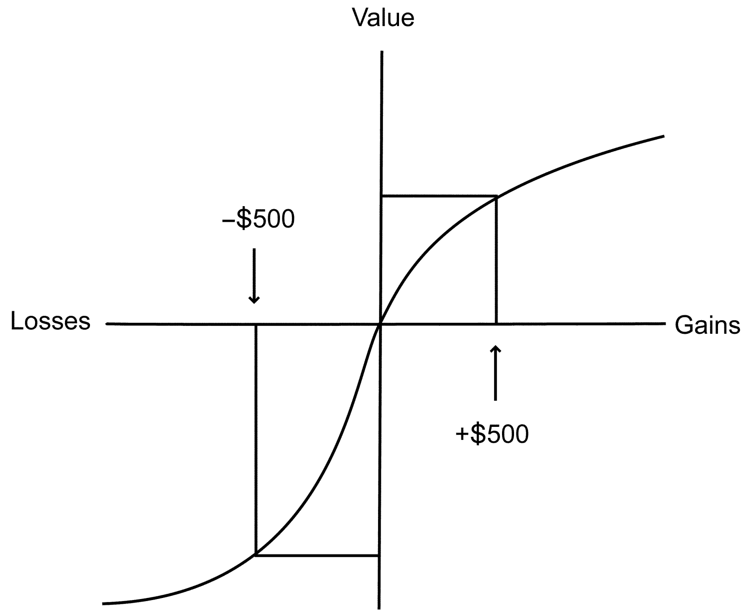


Figure 3.4. The values function in prospect theory (after Hansson, 1990). Here it is asymmetric, implying that a negative effect is more bad than a positive effect is good.

3.4. Maxiprobability

The maxiprobability decision rule makes the decision only on basis of the most probable outcome associated with each alternative. The rule requires criteria 1b and 2b in table 3.3 to be satisfied. The decision rule is insensitive to scenarios with low probability but high disutility, meaning that the biggest risks could be left out (Johannesson, 1998).

3.5. Minimax

The minimax decision rule (also named minimax regret, minimax risk, or minimax loss) focuses on the possible regret of each alternative and advices you to choose the option with the lowest maximal regret (to minimise maximal regret). This decision rule requires criteria 1c and 2a in table 3.3 to be satisfied.

3.6. Maximin and maximax

The maximin rule urges us to choose the alternative that has the maximal security level, thus the alternative that has the best worst outcome. Criteria 1c and 2b in table 3.3 must be satisfied. The maximax decision rule focuses on the best cases instead of the worst cases, thus choose the alternative whose best possible outcome is best. Same criteria as for maximin are required.

The maximin rule is commonly recognised as a major decision rule under uncertainty. The maximax rule however, is commonly regarded as highly irrational (Hansson, 1990; Johannesson, 1998).

3.7. Cost-benefit analysis

Cost-benefit analysis (CBA), or strictly *social* CBA was developed as a subject in order to be a practical guide to social decision-making (Brent, 1996). The decision rule is to choose the alternative that maximises the present value of all benefits less that of all costs. Criteria 1a and 2a in table 3.3 need to be fulfilled. Hence it requires a situation of complete knowledge. Of course, this is seldom true and CBA is often performed under risk or uncertainty, although this is not explicitly stated (Johannesson, 1998).

Brent (1996) breaks down the CBA process to four interrelated questions:

1. Which costs and which benefits are to be evaluated?
2. How are the costs and benefits evaluated?
3. At what interest rate are future benefits and costs to be discounted to obtain the present value (the equivalent value that one is receiving or giving up today when the decision is being made)?
4. What are the relevant constraints?

Brent (1996) then answers these questions from two different perspectives, namely from the perspective of maximising the welfare of a private firm and maximising the social welfare, table 3.4. It is a simple illustration of the different complexity in character of a "private" or social CBA.

Table 3.4. Answers to Brent's four questions from two different decision horizons (Brent, 1996).

	Social CBA	**Private CBA**
1.	All benefits and costs are to be included, consisting of private and social, direct and indirect, tangible and intangible.	Only the private benefits and costs that can be measured in financial terms are to be included.
2.	Benefits and costs are given by the standard principles of welfare economics. Benefits are based on the consumer's willingness to pay for the project. Costs are what the losers are willing to receive as a compensation for giving up the resources.	Benefits and costs are the financial receipts and outlays measured by market prices. The difference between them is reflecting the firm's profit.
3.	The social discount rate (which includes the preferences of future generations) is to be used for discounting the annual net-benefit stream.	The market rate of interest is to be used for discounting the annual profit stream.
4.	Constraints are not allowed for separately, but are included in the objective function. For example, income distribution considerations are to be included by weighting the consumer's willingness to pay according to an individual's ability to pay. A fund's constraint is handled by using a premium on the cost of capital, that is, the social price of capital is calculated which would be different from its market price.	The main constraint is the funds constraints imposed on the expenditure department.

3.7.1. Problem areas of CBA

There are some problem areas that arise in applying CBA to environmental issues. Hanley and Spash (1995) summaries the following:

(i) The valuation of non-market goods, such as wildlife and landscape. How should this be done, and how much reliance should society place on estimates so generated? Are we acting immorally by placing money values on such things?
(ii) Ecosystem complexity: how can society accurately predict the effects on an aquatic ecosystem of effluent inputs?
(iii) Discounting and the discount rate: should society discount? If so, what rate should be used? Does discounting violate the rights of future generations?
(iv) Institutional capture: is CBA a truly objective way of making decisions, or can institutions capture it for their own ends?
(v) Uncertainty and irreversibility. How will these aspects be included in a CBA?

Palmer (1998) acknowledges some difficulties to the use of economics as an appraisal mechanism, such as the distribution problem and more fundamental, the underlying assumptions of the approach of CBA. He concludes that it is limited in its application to situations of choosing between options "by the need to recognise the institutional, ethical and psychological problems, and by the need to make more complex analysis of behaviour in circumstances of risk and uncertainty." As suggestions to explicit solutions, Palmer (1998) mentions the inclusion of extending the economic model to include multi-attribute approaches or by limiting economics to being a source of information, leaving more explicit trade-off to political processes. He also states that in the appraisal there is undoubtedly benefit in using a consistent framework like that of CBA, and policy-making can only benefit from any process that makes more explicit the opportunity cost of taking one action or another.

Johannesson (1998) argues that CBA should only be regarded as an aid in decision-making since it does not solve the problem of uncertainty. He points out two main aspects that can be critical to the results of the analysis:

1. To put money on non-market goods.
2. How to discount future costs and benefits.

3.7.2. Valuation of non-market goods

A number of techniques have been developed for the valuation of non-market goods. The methods can be divided in direct and indirect methods. Indirect methods translate information about a certain market good that are related to the non-market good, which value is demanded. However, some values can not be related to any existing market good. For those non-market goods a hypothetical market can be put up, which is defined as a direct method. Table 3.5 summarises some methods for valuation of non-market goods. It is not possible to make a thorough description/explanation of the methods in this report. Hanley and Spash (1995), Brent (1996), Statens Naturvårdsverk (1997) and NRC (1997) give a good background to the different techniques.

Pearce and Seccombe-Hett (2000) list seven potential uses of environmental valuation.

i) CBA of projects.
ii) CBA of policies. (Since 1981, legislation in the United States has demanded that all new major regulations be subjected to CBA.)
iii) Pricing policy.
iv) Design of environmental taxes (ecotaxes).
v) National accounting – “green” accounting.
vi) As a management tool.
vii) As a participatory exercise.

The main uses of environmental valuation have, according to the authors, been for CBA, “green” national income analysis, and environmental tax design. Pearce and Seccombe-Hett (2000) present some practical experience with economic valuation as input to policy design, both at the pan-European level and at the national level within Europe, using the United Kingdom as an example. The authors claim that not accounting for economical valuation has imposed significant costs on Member States in for example the “excessive” standards for cleanliness of drinking water and bathing water and anticipate a greater future role for monetary valuation.

Table 3.5. Methods for valuation of non-market goods.

Indirect methods:	**Description**
Derived demand and production cost estimation techniques (NRC, 1997) or *dose-response functions* (Hanley and Spash, 1995).	These techniques impute a value of a non-marketed environmental input into a production process. Hence, these methods seek a relationship between environmental quality variables and the output level of a marketed commodity. The output may be defined either in quantity terms or quality terms.
Averting behaviour method (NRC, 1997) or the *avoided cost approach* (Hanley and Spash, 1995).	This method seeks the relationship between a change in environmental quality and household expenditures. The household may respond to increased degradation of consumption goods (such as water, noise etc.) in various ways that are generally referred to as averting or defensive behaviour.
Hedonic price/pricing method (HPM) (NRC, 1997; Hanley and Spash, 1995; Statens Naturvårdsverk, 1997)	The method seeks to find the relationship between the levels of environmental services and the prices of the marketed goods. This is typically applied to housing where house prices should reflect the capitalised value environmental quality to the house-owner.
Travel cost method (TCM) (NRC, 1997; Hanley and Spash, 1995; Statens Naturvårdsverk, 1997)	The idea behind the travel cost method is that people spend a certain amount of money on travelling to recreational areas such as national parks. This travel cost may be viewed as the price of access to the site. Under a set of assumptions it is possible to derive the individuals' demand for visits to a site as a function of the price of admission.
Direct methods:	**Description**
Contingent valuation method (CVM) (NRC, 1997; Hanley and Spash, 1995; Statens Naturvårdsverk, 1997)	The value is based on interviews where direct questions are asked of people’s willingness to pay (WTP) or people's willingness to accept (WTA) for a certain good. This good can be almost anything. The method has been applied to several ”goods”, such as; ”to save the Swedish wolf from extinction” or ”to reduce the nutrient outlet to the Baltic Sea by 50%”.
Conjoint analysis (NRC, 1997) or *contingent ranking method* (Hanley and Spash, 1995)	This method goes beyond the simple yes/no of a referendum format and asks individuals to reveal more information about their preferences by asking them to rank the hypothetical alternatives.
Contingent behaviour (or activity) method (NRC, 1997)	This method involves the use of hypothetical questions about activities related to environmental goods or services.

The Swedish Environmental Protection Agency (Swedish EPA) and other authorities have been commissioned by the Swedish government to propose intermediate goals in conjunction with the longer-term Swedish national environmental quality goals (Statens Naturvårdsverk, 1999a). The Swedish EPA recommended the sector authorities to use the National Audit Board’s (RRV) presentation model for material produced for socio-economic decision-making. The model of socio-economic assessment means weighting the costs and benefits of each measure or package of measures taken against each other. The socio-economic assessment may look different but some parts should always be included: (1) definition of the measure – zero-alternative and alternative measures, (2) identification of parties affected by the measure, both direct and indirect, and (3) identification of socio-economic costs and benefits. When possible, the following should also be included: (4) quantification of socio-economic costs and benefits, (5) monetary valuation of all relevant socio-economic effects,

(6) discounting of all costs and benefits, and (7) weighting costs against benefits (Statens Naturvårdsverk, 1999a).

Economical valuation of groundwater

The National Research Council (NRC, 1997) emphasises the importance of recognising and quantifying the total economic value (TEV) of a groundwater resource. The TEV is the sum of all services the groundwater resource provides. NRC (1997) divides them into two basic categories, extractive and *in situ* services. Extractive services are the municipal, industrial, agricultural and commercial demands met by groundwater. These values can be viewed as market goods and can more or less easily be calculated accordingly. *In situ* services are such values connected to the presence of groundwater in the aquifer, that the water remains in place. These may not be as obvious but are nonetheless important. *In situ* values identified by the NRC (1997) are listed below.

1) Groundwater may have ecological values as it may recharge watercourses and wetlands that have certain ecological functions.
2) A buffer value can be attributed for abstraction at special times.
3) Abstracting groundwater can cause soil subsidence that can be very costly; thus there is a (geotechnical) value of the groundwater if it remains in the ground.
4) In areas used for recreation groundwater has as a consequence of its ecological functions, recreational values.
5) In coastal area groundwater abstraction may cause sea water intrusion and hence degrade the aquifer for future use.
6) NRC also lists existence values, which is one of the most controversial topics in environmental economics. Existence value is a non-use function and purely the subjective satisfaction of knowing that there is unaffected water.
7) Bequest values are another non-use function. It is the value of leaving clean water to coming generations.

These last values, existence values and bequest values, are very difficult to estimate, but intuitively we may attribute them a value.

Examples of studies done in a societal perspective

Sandström (1998) proposed a framework for valuing the groundwater as a drinking water resource in a glaciofluvial deposit opposed to the value of exploiting it as a gravel resource. Three types of effects are suggested to be accounted for, changes in groundwater level and groundwater flow, changes of groundwater quality and changes in groundwater vulnerability (i.e. that a contaminant reaches the groundwater resource). Scenarios are created for different gravel exploitation alternatives and evaluated. The value of the glaciofluvial deposit is not only extractive values or as a gravel resource. Sandström (1998) points out that a proper valuation should include aesthetic, ecological, cultural, historical and recreational values, at least as a qualitative description.

Press and Söderqvist (1996) made a contingent valuation study in Milan where drinking water, narrowly meeting EU quality standards, is exclusively provided by groundwater extraction. The study was done to estimate the economical value of avoiding further degradation of groundwater quality. Gren, Söderqvist and Wulff (1997) conducted a study with the purpose to present the costs and benefits from reductions in the loads of nitrogen and phosphorus to the Baltic Sea. The study was carried out as an interdisciplinary cooperation among researchers from economics, geography, marine biology and ecology.

Environmental economics measure the monetary value of reduced mortality risk using the "value of a statistical life" (VSL) defined by individuals' preferences for small changes in risk and income. VSL is a measure of the value of small changes across a population and not how to value prevention of a specific death. Thus, if an individual's VSL is $5 million, he would pay $50 to reduce his risk of dying this year by 10^{-5}. The value of VSL does not in imply that the individual would pay $5 million to avert certain death this year, nor that the individual would accept certain death in exchange of that amount. It implies that 100 000 similar people would together pay $5 million to eliminate a risk that would be expected to randomly kill one of them this year. Hammitt (2000) reviews the theoretical foundations and empirical methods of estimated VSL and its dependence on variables such as age, income etc. Some empirical studies of the value of life saving is presented, based both on revealed and stated preferences. Estimates range between US$100,000 and US$10 million. Reviewers conclude that the most reasonable values are US$1.6-8.5 million (1986$) and later US$3-7 million (1990$). Very little correlation between stated WTP (by means of CV studies) and risk reduction was found in the reviewed literature. Hammitt (2000) points out that there is a need for research to clarify how individuals perceive differences in risk, in actual behaviour and in hypothetical settings. The difficulties in communicating hypothetical risk reduction to survey respondents lead according to Hammitt (2000) to varying estimates of VSL and diminish the credibility of CV studies.

Critics of CV studies
National Oceanic and Atmospheric Administration (NOAA) commissioned a panel due to the *Exxon Valdez* oil spill to judge whether CVM was a valid technique for measuring passive-use values associated with natural resource damage assessment. It resulted in a report that endorsed the method and set down a series of recommendations for conducting a CVM survey that were meant to be definitive (Spash, 2000). The Exxon case also produced a volume of studies critical to CVM and many of the recommendations of the NOAA panel. Some critics to the recommendations are:

- They were defined in a specifically American context.
- They recommend using willingness to pay (WTP) despite noting that willingness to accept (WTA) is theoretically required for damage assessment. The panel's argument that conservative values are in some sense preferred can compromise the valuations and distort incentives[6].
- How one should act when opposing parties have interest in the study.

Generally, the NOAA guidelines did include some sensible suggestions for survey conduct but also neglected a series of questions raised by psychologists among others (Spash, 2000).

Spash (2000) notes that protests among respondents in a CV survey might be expected on grounds of fundamental disagreement with the approach to choose between loosing environmental quality or paying some money to avoid it. This approach treats the environment as fundamentally identical to marketed goods and services. Spash (2000) refers to a number of studies where individuals have indeed been found to reject the idea of trading environmental quality and deny the principle of "gross substitution". Spash (2000)

[6] Early CVM studies took the position that WTP and WTA should give the same result. Empirical and experimental work however, showed that WTA format gave a proportional higher number of protest bids and that in most cases stated WTP was significantly lower than stated WTA. That WTA > WTP may be due to 1) loss aversion – individuals value a given reduction in entitlements more highly than an equivalent increase in entitlements. 2) Income constraints on WTP bids. 3) Risk-averse consumers that are given only one chance to value the good will on average overstate WTA and understate WTP (Hanley and Spash, 1993).

investigates the ethical variables among the respondents and conclude that the monetary values obtained fail to represent the exchange prices and welfare changes which economists are trying to derive.

Sagoff (2000) argues that existence values (only measurable with CVM studies) reflect what people think society ought to do not what they believe will benefit them. When, in essence, the economic perspective is to reduce the public good to the welfare of the individual, this is not as Sagoff (2000) states "a perspective particularly congenial to the common-law tradition, the religious history, and the intellectual heritage of our nation."

3.8. Decision criteria for uncertainty

Hansson (1990) presents five different decision criteria for decision-making under uncertainty. Here two are mentioned. Several of the most discussed decision criteria under uncertainty try to compromise between these two. They require some kind of measure of uncertainty of the probability of the different outcome (see further part 4.1).

Maximin expected utility (MEU)

The decision rule is to choose the option with the largest minimal expected utility.

Reliability-weighted expected utility

It is the same as the MEU decision rule with the addition that the probabilities are reliability-weighted and thereafter used as a probability value.

3.9. Social decision theory

Social decision theory is concerned with the aggregation of individual preferences (Hansson, 1990) The aim is to find a rational way to combine individual preferences into a social choice. It is thus a collective decision theory. The most common decision rule is the majority rule. However, as Hansson (1990) shows this may result in decision instability.

4. UNCERTAINTIES

4.1. Introduction

Decision situations at contaminated sites are in practice risk management situations. They represent situations under uncertainty where the uncertainties at hand can be of various kinds. Some of them we will describe in numbers, some of them we will not. And this is of course one of the more problematic things in decision situations, the aspects that we do not consider at all. Nevertheless, it is of importance to try to describe the uncertainties at hand. Issues to consider are for example uncertainties of the geology and hydrogeology at the site, uncertainties of the amount and degree of contamination, uncertainty how the contaminant is transported, what the concentrations will be at the point of exposure to targets, and how it will affect the identified targets. These uncertainties can be described in numbers, whereas uncertainty to for example what the actual targets are is more seldom described in that way. This last example can be seen as a model uncertainty, which will be discussed more in part 4.3. Worth discussing is how we choose to describe these uncertainties and how we estimate them. In this part, an overview of uncertainties of different kind is presented.

Uncertainty and variability

It is in most of the literature differentiated between uncertainty and variability. Uncertainty is due to the assessor's perception of the system. Variability on the other hand is due to the heterogeneity of that system. Variability and uncertainty is not always separated in quantitative estimates of uncertainty. For example, Freeze et al. (1990) uses the ergodic hypothesis[7] to combine variability and uncertainty in their calculation of the hydrogeological risk. This treats the system as homogeneous but random.

The Committee on Risk Assessment of Hazardous Air Pollutants (NRC, 1994) prefer to use the concepts of parameter uncertainty and model uncertainty as opposed to variability and uncertainty. Uncertainty in parameter estimates arises due to measurement errors, use of generic data, or non-representativeness to mention a few. Model uncertainty arises due to gaps in the scientific theory.

Uncertain probabilities

Hansson (1990) defines risk as complete probabilistic knowledge and uncertainty as partial probabilistic knowledge. Thus, when a risk is at hand we would know the exact probability for the event to occur. When there is an uncertainty at hand the knowledge of the probability for the event is only partially known. Hence, there is a degree of uncertainty of that probability. There are methods to express the incompletely known probabilities such as the binary measure that divides the probability values in two groups: possible and impossible values. Another way of describing it is by using a second-order probability, thus a reliability measure of the probability to have a certain value. To describe a probability for an event with the use of a probability density function is a way of expressing a second-order probability. Further Hansson (1990) mentions Fuzzy sets memberships as a way to express the vagueness that is the uncertainty about a probability. In practice, all situations at contaminated sites are situations under uncertainty. Mostly however, it is treated as situations under risk, where it is assumed that all probabilities can be correctly described.

[7] "*Ergodicity* implies that the unique realization available behaves in space with the same pdf (probability density function) as the ensemble of possible realizations. In other words, by observing the variation in space of the property, it is possible to determine the pdf of the random function for all realizations." (de Marsily, 1986)

4.1.1. Estimations of uncertainties

In risk assessment expert judgement, traditional statistics and many other methodologies are used for the quantification of a certain probability. Richards and Rowe (1999) devoted an article in *Risk Analysis* to the subject of decision-making with heterogeneous sources of information. That is, sources with different levels of uncertainty. Generally, they identify four different types of uncertainties: temporal – uncertainty in future and past states, structural – uncertainty due to complexity, metrical – uncertainty in measurements and translational – uncertainty in explaining uncertain results (Rowe, 1994; Richards and Rowe, 1999). Rowe (1994) discusses these different dimensions of uncertainties and their parameters. Often the variables in a decision model are made up with various sources of information and hence different types of uncertainties with different possibilities to validate. Richards and Rowe (1999) presents a hierarchy of different sources of uncertainty based upon ones' ability to validate data and models empirically: (1) standard distribution, (2) empirical distribution, (3) validated model, (4) unvalidated model, (5) alternate models, (6) expert value judgement, (7) best guess estimate, and (8) test case. The authors present a number of approaches to address the problem and point out the importance to evaluate what one do not know and to make this explicitly visible to the evaluator of the results.

Bayesian statistics can be used to formally use both subjective and objective information often called soft and hard data. Bayes' theorem is used to update our present model (our prior subjective estimate of the model) of the true state of the system by means of additional information (e.g. sampling). In words (Alén, 1998):

$$\left(\begin{array}{l}\textit{Posterior}\\ \textit{probability of the}\\ \textit{true state } \theta \textit{ given}\\ \textit{the sample}\end{array}\right) = \left(\begin{array}{l}\textit{Norma-}\\ \textit{lising}\\ \textit{constant}\\ \end{array}\right) \cdot \left(\begin{array}{l}\textit{Sample}\\ \textit{likelihood}\\ \textit{given the true}\\ \textit{state } \theta\end{array}\right)\left(\begin{array}{l}\textit{Prior}\\ \textit{probability}\\ \textit{of the true}\\ \textit{state } \theta\end{array}\right)$$

There are studies done to investigate how subjective estimates of probabilities can be more reliable and how they affect decisions. Keeney and von Winterfeldt (1991) present the result of a study done to improve the use of expert judgements in complex technical problems. Instead of as often is the case an informal, implicit and undocumented use, they propose a formal way of elicitation of expert judgement. Hammitt and Shlyakhter (1999) investigates how prior (subjective) estimates of probability distributions influence the decision on whether or not to collect additional data, using the concept of expected value of information (EVI[8]). They conclude that if the prior distribution is too narrow, the calculated EVI will be biased, and that it is likely to be a downward bias in cases of interest to risk analysis. The downward bias is caused by the tendency of individuals to be overconfident in summarizing their information, and especially to underestimate the probability of surprise (see also Johannesson, 1998). The authors recommend to adopt long-tailed distributions to reduce such underestimates. As Taylor (1993) notes "…if the prior distribution is characterized by probability highly concentrated on a narrow range of values, it is less likely to be influenced by experimental data." The phenomena of overconfidence in subjective estimates is well-known as noted also by Hansson (1990) and Hammitt (1995). Hammitt (1995) notes that because of overconfidence in prior probability estimates, additional information can actually increase the uncertainty. Olsson (2000) investigates different methods of practically estimating uncertainties as unbiased as possible.

[8] Studies on data worth analysis is shortly presented in parts 5.1.1 and 5.3.5.

Taylor (1993) examines how probability distributions can be developed for model input variables in Monte Carlo simulations based on objective and subjective information. The author states that input distributions are developed somewhat haphazardly and means that there is a need for doing this more consequently. Often the combined use of empirical evidence and subjective information is preferred. The selection of distributional family (uniform, normal, Student's t, etc.) is based on how the distribution will be used and the fit of data. A simple approach to identify influential inputs prior to simulation is given. It can also be used to assess how large the uncertainty or variability in a parameter would have to be to influence the output distribution appreciably. Taylor (1993) considers exposure via ingestion of contaminated food:

$$Exposure = \frac{Cp \cdot M \cdot Ed}{Bw \cdot At}$$

where Cp is the concentration of contaminant in tissue of consumed plant crop, M is the consumption of locally grown crop, Ed is the exposure duration, Bw is the body weight, and At is the averaging time of exposure. Exposure is expressed in units of mg/kg/day.

If the inputs in the equation are independent then:

$$\sigma^2{}_{\ln(Exp)} = \sigma^2{}_{\ln(Cp)} + \sigma^2{}_{\ln(M)} + \sigma^2{}_{\ln(Ed)} + \sigma^2{}_{\ln(Bw)} + \sigma^2{}_{\ln(At)}$$

where σ^2 is the variance. The fraction of the variance in the natural log of exposure attributable to each input, x_i, is consequently $\sigma^2{}_{\ln(xi)}/\ \sigma^2{}_{\ln(Exp)}$. The calculation can be carried out if the x_i are lognormal, loguniform, or logtriangular or can be approximated accordingly (Taylor, 1993).

4.2. Uncertainties associated with the geological system

Wood (2000) writes in his editorial note in *Ground Water* that heterogeneity was first mentioned in the late 1950s and early '60s as a result of aquifer contamination forcing hydrogeologists to look at smaller aquifer volumes. What follows from this statement is not only the need for being able to describe the heterogeneity but also to adapt the parameter statistics to the scale of interest in a study. Today there is a wide recognition of heterogeneity and different approaches to deal with it.

Geostatistics is the statistics of observations located in space or time that can be correlated spatially or temporally (Hohn, 1999). It provides a set of tools to describe how geological parameters and geological stratigraphy vary spatially by different interpolation techniques (different types of kriging). It can also be used to quantify local uncertainty. Dealing with spatial data, the main difference between kriging (geostatistics) and Bayesian updating is the use of subjective prior estimates. The use of prior estimates seems to be relatively widespread in recent literature.

Freeze et al (1990) distinguish between two uncertainty models that should be used to describe the total uncertainty in a hydrogeological simulation model (see further part 5.3): geological uncertainty model and parameter uncertainty model. Geological uncertainties are identified as uncertainties associated with the thickness of geological layers, hydrogeological boundaries etc. Parameter uncertainty describes the variability of parameters such as hydraulic conductivity, porosity etc, see figure 4.1. A lot of research is devoted to studies on parameter uncertainty, especially the hydraulic conductivity (see the following part).

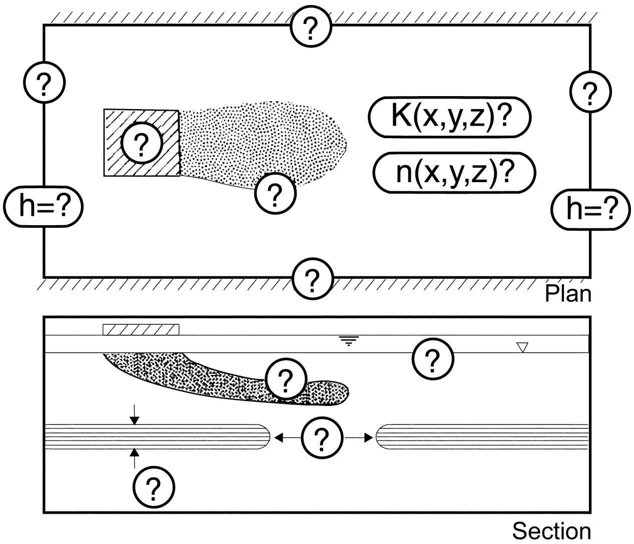


Figure 4.1. Geological uncertainties are uncertainties associated with the thickness of geological layers, hydrogeological boundaries etc. Parameter uncertainty describes the variability of parameters such as hydraulic conductivity, porosity etc. After Freeze et al. (1990).

4.2.1. Examples of studies on parameter uncertainty in geological data

Levy and Ludy (2000) used a Gauss-Hermite quadrature approach to quantify the uncertainty of the delineation of one- and five-year wellhead protection area (WHPA) for two municipal wells in a buried-valley glacial-outwash aquifer (not including spatial correlation). A shallow, unconfined flow system was modelled with MODFLOW, where six modelling parameters were used for the uncertainty analysis. The horizontal hydraulic conductivity of the outwash deposits, vertical-to-horizontal hydraulic conductivity anisotropy ratio for the outwash deposits, horizontal hydraulic conductivity of the silty-sand, recharge, river conductance and effective porosity of the outwash deposits. The prior estimates of each parameter were derived from field data, literature data, other modelling studies and subjective hydrogeological understanding and then modified based on the study's MODFLOW simulations. The flow model was used to explore the ranges of possible parameter values that still produced acceptable calibration results given the possible ranges of all other model parameters included in the uncertainty analysis.

Copty and Findikakis (2000) estimated the uncertainty quantitatively in the evaluation of groundwater remediation schemes due to natural heterogeneity represented by the hydraulic conductivity. The hydraulic conductivity was defined as a random spatial variable whose statistical structure was inferred from available hydraulic conductivity data. Multiple realisations of the hydraulic conductivity field was generated by Monte Carlo simulations. The probability that each of the realisations may represent the actual hydraulic conductivity field was estimated by simulating the historical spread of a groundwater plume and compared with measured concentrations. Bayes' theorem was used to produce a conditional probability of each realisation. The proposed remediation alternatives was simulated for all realisations and the output weighted by the conditional probability of the hydraulic conductivity field realisation. Finally, the performance of each remediation alternative was evaluated statistically. A numerical example was used to demonstrate the importance of modelling probabilistically rather than deterministically because of the biased results that may be obtained from a deterministic model.

Dou et al (1997) developed a method to incorporate ”soft” information in a transient groundwater flow simulation using a fuzzy set approach. “Soft” or imprecise data may be indirect measurements, expert judgement, subjective interpretation of available information etc. The Fuzzy sets were used to describe imprecision (vagueness) in a nonprobabilistic framework and the output of the simulations resulted in direct representation of uncertainty, in this case hydraulic head uncertainty (Dou et al., 1997).

The uncertainty in predicting the rate of mass removal created by soil vapour extraction (SVE) systems in the unsaturated zone due to soil heterogeneity was investigated by Barnes and McWhorter (2000b). It was noted that SVE systems have consistently shown an initial large mass removal in a short time and that the rate of mass removal decreases sharply as the removal process continues. This is due to diffusion, which is the dominant mechanism in the removal of compounds from regions of low permeability and high soil-water saturations in heterogeneous soils. Barnes and McWhorter (2000b) developed a method to represent the heterogeneities in permeability using Monte Carlo analysis and use the model to illustrate the effect on mass removal predictions. The authors conclude that a deterministic modelling of a SVE system may fail to predict the rate of mass removal.

Abbaspour et al. (1998) present a model of uncertainty analysis, BUDA (Bayesian Uncertainty Development Algorithm) to account for the special characteristics of environmental data: spatial and/or temporal autocorrelation, natural heterogeneity, measurement errors, small sample sizes, and simultaneous existence of different types and qualities of data. They treated hydraulic conductivity, porosity and longitudinal dispersivity as random variables in order to model a chloride plume from a landfill. The prior uncertainties of these variables were subjectively estimated. The uncertainties are propagated to a goal function, which defines the best alternative. A data worth model was used for the reduction of uncertainty in the model.

4.3. Uncertainties in conceptual models

Clearly, the understanding of the different processes such as release, transport, and exposure is the base for a risk assessment. Conceptual models are often considered as an important part for developing this understanding. Gorelick (1997) points out that going from observations of the true system to the conceptual model is the most crucial step in simulation model development.

Olsson et al. (1994) differentiate between a theory which is expected to be generally applicable and a model which is used to provide a representation of a process or system for a specific purpose. The model generally attempts to describe the aspects of nature that we think are important for the problem we attempt to solve or the prediction we attempt to make. If the approximations introduced in the definition of a model is valid or not can only be judged in relation to the purposes of the application.

In relation to hydrogeological simulation models, Olsson et al. (1994) proposes that the term conceptual model should be "…restricted to define in what way the model is constructed, and that this is separated from any specific application of the conceptual model.” A conceptual model is proposed to define the geometric (or structural) framework in which the problem is to be solved, the size of modelled volume (scale), the constitutive equations for the processes included in the model, and the boundary conditions (Olsson et al., 1994, see figure 4.2). Dagan (1997) points out in relation to stochastic modelling of groundwater flow that it is important to specify the aim of the modelling effort because the models are problem oriented.

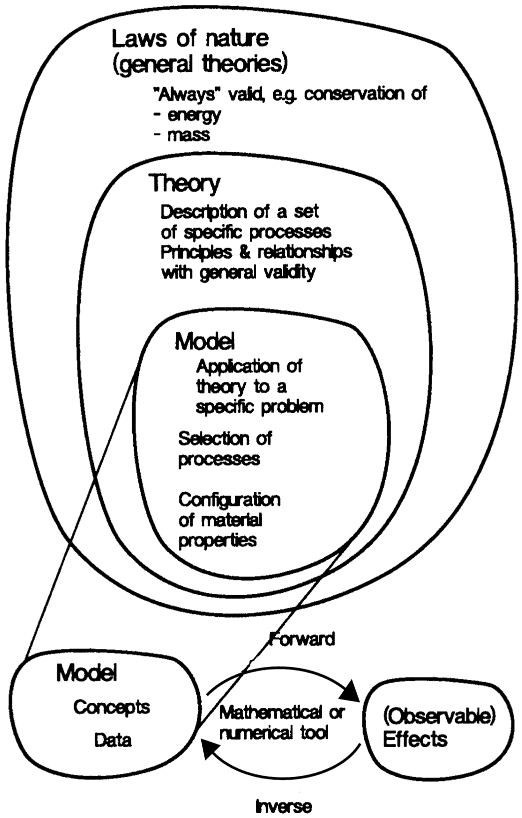


Figure 4.2. Hierarchy of theories and models. After Olsson et al. (1994).

A typical conceptual site model (CSM) in relation to risk assessment should according to Asante-Duah (1998) include the following basic elements: 1) Identification of site contaminants and determination of their physical/chemical properties. 2) Characterisation of the source(s) of contamination and site conditions. 3) Delineation of potential migration pathways. 4) Identification and characterisation of all populations and resources that are potentially at risk. 5) Determination of the nature of inter-connections between contaminant sources, contaminant migration pathways, and potential receptors. An accurate design is one that will meet the overall goals of a risk assessment and environmental management program. Additional and new data necessitates a re-design, or updating of the CSM.

American Society for Testing and Materials (ASTM, 1995) developed a standard guide for developing conceptual site models for contaminated sites. The document guides through activities such as; assembling information, identifying contaminants, establishing background concentrations of contaminants, characterising sources, identifying migration pathways and identifying environmental receptors. It is pointed out that the complexity of the model should be consistent with the complexity of the site.

The definition of a conceptual model is more or less exact with different authors, but there seems to be a general agreement. Two things are worth noting; a conceptual model is only valid for a specific purpose and the process of developing a conceptual model should be iterative and based on scientific reasoning, considering both available data and information.

4.3.1. Quantification of model uncertainty

Generally, model uncertainty is seldom accounted for. Dagan (1997) points out that "…assigning probabilities and incorporating conceptual models in a formal, quantitative framework has not been given sufficient attention in the literature, and generally only one such model is chosen by modellers."

A study by Bethke and Brady (2000) compares the use of the distribution coefficient (K_d) and the use of surface complexation theory in contaminant (Pb) reactive transport models used for example to design remediation schemes. The result of the two different approaches is qualitatively different. Surface complexation theory is more realistic physically and chemically but requires much more data input in the model. The authors conclude that the results are different enough to make it worthwhile to use surface complexation theory. They do not, however, quantify the effects in terms of for example costs.

Russell and Rabideau (2000) have another approach when examining different modelling assumptions. They use two single-layer conceptual models of different complexity in combination with different degrees of aquifer heterogeneity (variance of ln[K]) to model a pump-and-treat design. They use the results in a decision analytical framework to assess the impacts of the different assumptions. Whereas the assumed aquifer heterogeneity had a large impact, the impact of complexity of the single-layer model was less.

4.4. Uncertainties in risk assessment

Risk assessment is associated with many uncertainties, not just hydrogeological uncertainties but also uncertainties in toxicological data, cancer potency factors etc. Although this topic is somewhat outside the scope of this literature review, it should be worthwhile to mention some approaches. Primarily though, it is important to point out that it is not only the geology which is heterogeneous but that populations are so as well. As Hamed (1999) points out, there has been a growing trend towards using probabilistic methods in ecological and public health risk assessment. The reason for this is that stochastic methodologies circumvent the need to use single values that are usually overly conservative (Batchelor et al., 1998; Hamed, 1999).

As examples of variability in risk assessments the Committee on Risk Assessment of Hazardous Air Pollutants (NRC, 1994) present: emissions variability, atmospheric process variability, microenvironmental and personal-activity variability, and variability in human susceptibility. The upperbound point risk estimates typically computed by the US EPA does not convey the degree of uncertainty in the estimate, which causes the decision-maker to be ignorant of the extent of conservatism, if any, that is provided in the risk estimate. NRC (1994) recommend a formal uncertainty analysis as an iterative process where "the health or economic impacts of the regulatory decision are large and when further research is likely to change the decision."

Hoffman and Hammond (1994) argue that few risk assessments performed contain a formal uncertainty analysis and if it does, it is often qualitative although a quantitative approach would be more defensible. They identify the need to carefully define the endpoint or target of the risk assessment since this will influence the method of uncertainty analysis. Two types of uncertainties are described, Type A and Type B in accordance to Safety Series No. 100 of the IAEA (International Atomic Energy Agency). Uncertainty about a quantity that is fixed (deterministic) with respect to the assessment end point is called Type B uncertainty. When the assessment end point is a distribution of actual exposures or risks (but the exposure to specific individuals in the population remains unknown), the uncertainty is of Type A. It is thus a separation of uncertainty and variability.

4.4.1. Examples of studies on uncertainty in risk assessment

Maxwell and Kastenberg (1999) made a stochastic study on predicting cancer risk from contaminated groundwater. The model included heterogeneous subsurface flow and contaminant transport, subsurface receptor locations (wells), a water distribution network, and a household exposure model. In the household individuals may be exposed to the contaminated groundwater by means of various household activities, such as washing, drinking and showering. The methodology includes uptake, metabolised dose and estimation of low-dose effects. They used a nested Monte Carlo approach to separate between uncertainty and variability. All parameters are assumed to incorporate some elements of uncertainty and variability. The geological heterogeneity, represented as spatially variable hydraulic conductivity, influences uncertainty and variability in groundwater flow and contaminant transport. Representation of the uncertainty in cancer potency, and the uncertainty and variability in rates of metabolism of carcinogenic compounds and in the hydraulic conductivity is important when predicting human health risk.

Batchelor et al. (1998) conducted a stochastic risk assessment at a PCB contaminated site in Texas. Four pathways were considered: inhalation, dermal contact, ingestion and consumption of fish or cattle that had been exposed. All variables in the risk assessment model were given a probability density function (PDF) and simulated in Crystal Ball with the Latin Hypercube sampling method with 100,000 iterations. The upper bound of total cancer risks were estimated to be in the range from 10^{-5} to 10^{-4}. The stochastic calculations were compared with deterministic conservative calculations which correspond to the 99.7-99.8% points of the distribution for risk calculated by the stochastic model. Preliminary remediation goals were calculated at different risk assurance levels.

Hamed and Bedient (1997) investigated the performance of computational methods for risk assessment from PCE contamination of a groundwater supply in California. Monte Carlo simulation was compared with first- and second-order reliability methods. For the first case study, PCE concentration in the water was obtained from sampling from the water supply and the result was investigated for a number of different target risk levels. Exposure pathways included was dermal contact with water, ingestion and inhalation with nineteen different variables. The variables with most impact on the result was dependent on the risk target level but generally PCE metabolised cancer potency and PCE concentration in water. As a second case study they consider the risk due to ingestion only of benzene contaminated groundwater where they include a groundwater transport model. Where uncertainty in fate and transport parameters was included, these had the most impact on the probabilistic outcome (decay coefficient and contaminant velocity, including the retardation factor).

Hamed (1999) made a probabilistic sensitivity analysis using the first-order reliability method of the cancer risk resulting from dermal contact with benzo(a)pyrene (BaP)-contaminated soils. Nine parameters were considered random: average body weight, fraction of skin area exposed, soil concentration, cancer potency factor, average body surface area, skin soil loading, time soil stays on skin, soil bulk density, and skin water content. On the other hand, variables such as organic carbon fraction, soil water content, and soil porosity was chosen deterministically. Next, Hamed (2000) investigated the effect of the choice of PDF on the probabilistic outcome in the above mentioned study. The distributions for the variables cancer potency factor, soil concentration and fraction of skin area exposed was investigated as these had the most impact on the outcome. Three different distributions were considered: normal, lognormal and uniform. On the 50th level for probability of failure for incremental lifetime cancer risk the PDFs of the studied variables had only moderate impact. On the 95th level on the other hand, the impact was much greater. Hamed (2000) conclude that the chosen PDF will generally have a large impact in public health risk assessment because in most cases the regulatory risk is at the tail end of the risk distribution.

4.4.2. Uncertainty in environmental standards

Risk assessment can be used to derive generic guidelines. As discussed, the models can incorporate several uncertainties. According to Whitehouse and Cartwright (1998), Barnett and O'Hagan [1997] argue for "statistically verifiable ideal standards", which combine a level that should not be exceeded with a standard for ensuring statistical verification of compliance with that limit. Whitehouse and Cartwright (1998) conclude that: "More robust standards would result from an explicit recognition of variability, for example by expressing standards as probability density functions rather than oversimplistic threshold concentrations. This would also enhance the utility of standards because it opens up the possibility of probabilistic risk assessment and more effective cost-benefit assessment." Andersson (1999) applies this view, referring to the statistical verification as a specified risk acceptance level. Because, as Andersson (1999) notes, a deterministic regulating framework is insufficient for controlling and managing risks (risk is defined as a probability of event).

Freeze and McWhorter (1997) developed a framework to investigate the risk reduction due to DNAPL mass removal from low-permeability soils. Regulatory compliance in most cases takes the form of maintaining dissolved trichloroethylene (TCE) concentrations below specified standards at compliance monitoring wells. They presume an alternate concentration limit (ACL) back calculated from a maximum contaminant level or from a health-risk characterisation. The reason is that it is risk-based and thus allows for direct consideration of risk-reduction. A tutorial, prepared by Sandia National Laboratories (1998) demonstrates the use of SmartSampling at a Mound Plant in Miamisburg, Ohio. SmartSampling is developed for the purpose of environmental decision-making at contaminated sites. The defined decision rules for the remediation are: all soil will be removed where the probability of exceeding 75 pCi plutonium/g is greater than or equal to 0.05, and all soil will be removed where the probability of exceeding 150 pCi plutonium/g is greater than zero. The rule is formulated in a probabilistic way to account for uncertainties.

5. DECISION-MAKING FRAMEWORKS APPLIED AT CONTAMINATED SITES

Regulations and guidelines have been set up through out the different countries. These have developed as more and more is learnt about contaminated sites and contaminated groundwater. There are in principle two ways of looking at regulations. Either we will approach it by cleaning up the site until the allowable concentration is reached. A second approach is to have a risk-based view of the problem. By a risk-based view is here meant to find an acceptable equilibrium between risks, uncertainties and costs in order to allocate resources in a proper way.

A crucial question is how to find this equilibrium. When do we have acceptable risks, when are the risks unacceptable? To which cost can we reach an acceptable risk? Is it worth it? Are there alternative ways to obtain this level of ambition? Many scientists have been busy to try to structure frameworks in helping answering some of these questions and there are many different approaches to deal with it. The following chapter will try to present some of the work done in this field.

There are many suggestions in the literature on how to structure information such that a decision can be made concerning strategies at contaminated sites. The previous parts have focused in the background of the problem whereas this part will focus on different author's suggestions on how to solve it. The most used decision rule is the expected utility rule. Pure Bayesianism is not found where both probabilities and utilities are purely subjective. Instead, objectivist EU is commonly extended with the use of subjective estimates of objective probabilities, thus Bayesian updating of probabilities and parameters. In applications of data worth analysis, Bayesian updating plays a crucial role.

The degree of knowledge applied in the decision situation is mostly considered to be a complete probabilistic knowledge, thus situations under risk. However, since data worth analysis is often suggested, the degree of knowledge is implicitly under uncertainty. Some of the authors suggest to include the probability density function (or distribution of the probability) instead of a point estimated probability. This can be seen as an explicit way to express a variation of the probability, thus expressing that the decision is being made under uncertainty.

First, the difference between optimisation and decision analysis is shortly described in part 5.1 and a few early examples given of its application of data worth analysis and decision analysis. This is because the basic idea of a present uncertainty and the use of Bayesian updating are used in later applications of decision analysis. Second, suggested frameworks not based on optimisation before 1990 are described in part 5.2. The reason for this is due to an extensive work by Freeze and co-workers in 1990-1992, which summarised and extended most of the previous ideas. This work is shortly described in part 5.3. Some case studies and variations of this framework are also presented in this part. Finally, some suggestions of social decision theory are described in part 5.4.

5.1. Optimisation

Freeze and Gorelick (1999) made a comparison between stochastic optimisation and decision analysis for the design of remedial pump-and-treat systems in contaminated aquifers. They point out the main similarities and differences of the both decision-making frameworks. Additionally they propose possible ways to combine the two methods. The fundamental difference lies in the fact that decision analysis considers a broad suite of technological strategies from which one of many predetermined design alternatives is selected as the best and stochastic optimisation determines the optimal solution of a single technological strategy at a time. Thus solutions of decision analysis are not truly optimal because the discrete number of design alternatives are not likely to include the optimal one. On the other hand, decision analysis has no problem with the consideration of non-linear and discontinuous problems.

Optimisation allows identification of truly optimal values of the decision variables (for example pumping rates) and their continuous range. Linear programming is the most developed technology. It requires all functional relationships within the framework, namely the objective function and constraints, to be linear. If any non-linearity exists in the system it requires the use of non-linear programming, which requires more computational efforts. Discontinuities can cause large problems and recent advances in simulated annealing, neural networks, and genetic algorithms are being developed to deal with such difficulties (Freeze and Gorelick, 1999).

There are different ways to view the problem. Deterministic optimisation gives rise to a single-valued deterministic optimum, whereas stochastic optimisation provides a probabilistically distributed stochastic optimum being a function of the reliability. Often this is presented in the form of a trade-off curve that allows the decision-maker to choose degree of reliability in relation to the cost. Deterministic or stochastic optimisation can use either linear or non-linear programming. One may also achieve single-valued optimum solutions with stochastic optimisation by including a defined level of reliability.

There are many studies done in the field of optimisation. For example, Mylopoulos et al. (1999) used a stochastic optimisation approach for designing pumping wells for the purpose of aquifer remediation. The location of the wells was fixed whereas the pumping rate was optimised as to minimise the total operating cost. However, a full review of the theory and different case studies will not be included in this literature review. Instead, the main interest is on the relation between optimisation and decision analysis, about which Freeze and Gorelick (1999) has made an excellent contribution. Below are a few early examples that are of interest in the light of decision analysis.

5.1.1. Early works on decision analysis frameworks with optimisation

Haimes and Hall (1974) present a multi-objective decision framework to solve the problem of noncommensurable objectives, thus objectives that are not possible to reduce to a single measure. The basic principles are an optimisation framework combined with the construction of surrogate worth trade off functions to weigh the different objectives against each other. First the objectives are identified and optimised separately. Thereafter a trade off matrix is constructed based on asking the decision-maker of his or hers preferences. This can be done in some different ways proposed by the authors. The trade off matrix is then used to provide the best solution given the multiple objectives.

Kaunas and Haimes (1985) combine risk and optimisation for risk management of groundwater contamination in a multi-objective framework. They present a hypothetical problem that considers three objectives: to minimise the cost of prevention of contamination, to minimise the proportional time of contamination and to minimise the sensitivity of the contamination time to uncertainties in dispersivity. The method is intended to optimise the measures taken to reduce the threat of random industrial solvent (here: trichloroethylene, TCE) spills into an aquifer used for drinking water. The following models and methodologies are used: 1) a mass transport model to find well solute concentration response to an impulse spill input and 2) random number simulation to create a stochastic time series of spills and to find pollution time for chosen values of decision variables. Further: 3) linearity/convolution to obtain data on concentration versus time in well water, 4) regression analysis is used to compute a functional form of the pollution time ratio objective [$f_2(x)$] versus investment decision variables and dispersivity, and 5) risk dispersion index method (RDIM) is used to calculate the sensitivity index of $f_2(x)$. This index approximates the standard deviation of the contamination ratio $f_2(x)$. Finally, 6) trade-off analysis is used to find a preferred solution to a multi-objective optimisation problem using the concept of trade-offs among multiple objectives, as in Haimes and Hall (1974).

Jettmar and Young (1975) use economic results derived from optimisation of reservoir size to compare the use of simple or complex synthetic hydrologic data generator to historic data as input in a reservoir model. Thus instead of using the decision model to answer the question of how to design a reservoir, they use the decision model as a means to investigate different ways of generating data.

Marin (1986) and Loaiciga and Mariño (1987) made similar studies on how different methods of parameter estimations affect management decisions. The economical model is in both papers a minimisation of a loss function. Marin (1986) uses a general assymmetric loss (or regret) function, whereas Loaiciga and Mariño (1987) uses a quadratic and symmetric loss function.

Marin (1986) compares decisions made by classical parameter estimations (maximum likelihood techniques) with those made by Bayesian parameter estimation. The Bayesian parameter estimation procedure takes into account both the uncertainty of the parameter as well as the (economical) loss that can arise due to a non-optimal decision. Although in the examples given, the Bayesian approach results in more efficient decisions, the general conclusion is that the incorporation of parameter uncertainty does not guarantee improved management decisions, depending on the formulation of the loss function.

Loaiciga and Mariño (1987) made their study against the background that simulation models was being more incorporated in management schemes and that it had previously been shown that the statistical properties of parameter estimates affect the simulated field variable. Thus, it could be expected that different approaches for parameter estimations lead to different management solutions, which is shown in their study. They compare the classical (non-random but unknown), the Bayesian (random), and the deterministic (fixed and known) approaches for parameter estimation.

5.1.2. Early works on data worth analysis

Generally, data worth analysis considers the evaluation of reduction of uncertainty of additional data *before* the data is being measured. The worth of additional data can be viewed differently. The aim can be to reduce uncertainty and the best additional data is that which reduce the uncertainty as much as possible. Another viewpoint is to consider the worth of additional data as a function of not only the system that is described, but also of the economic importance of the decision being made. Thus, if the additional data is more costly in terms of change of the outcome of the updated economical objective function, then the new data is not worth its price. Another criterion for additional information to be of any value is if the information will change the decision. If the new data does not change the decision, then it has no value.

Davis and Dvoranchik (1971) consider additional information valuable only if its possession may cause a change of decision or action. The problem at hand is to decide whether another year's peak flow data in a river is worthwhile when designing a bridge. The variable cost of the bridge (Bayes' risk) is depending on both the construction (pile depth) and a risk-cost if the bridge is lost during a flood. The expected variable cost is calculated as a function of pile depth, and the pile depth that minimises the expected total cost is sought. This optimum pile depth however, is based on limited data. The crucial question therefore is to decide whether another year's data reduce the cost of the bridge design enough to wait for this year's data.

The authors use a nice little story to demonstrate the basic idea of the data worth analysis. To see how good their prior selected design is, they set up a meeting with Professor I. M. Clairvoyant who can tell them the true yearly peak flow distribution. Using this they may calculate an opportunity loss, OL (or regret) namely the loss of not selecting the true best design. This is also the maximum consulting fee they are willing to pay the Professor (the Value of Perfect Information, VPI). The Professor's crystal ball was fractured and the authors solve this by calculating the expected opportunity loss (XOL) instead. This is the same as the Expected Value of Perfect Information (EVPI). To calculate the value of next years data they want to consult the Professor again but unfortunately, he removed all activities from campus and the authors must therefore estimate it instead. They calculate the Expected Value of Sample Information (EVSI) as the difference of XOL and the expectation of next year's expected opportunity loss given one more sample (XXOL). Thus:

EVSI = XOL – XXOL

Davis et al. (1972) further examines the use of data worth analysis in hydrologic design, i.e. flooding protection measures. The basic idea is the same using Bayes' theorem when updating their knowledge. Grosser and Goodman (1985) describe a methodology to determine groundwater sampling frequencies (samples per year) by means of Bayesian updating. They use Bayes' theorem for continuous probability distributions and solve the updating process numerically. They point out that the objective (or goal) function should be developed such that it represents the true cost to an owner or to society.

Maddock (1973) sketches a framework for optimising the management of an irrigated farm. A data worth analysis is done by ranking the parameters that are the most critical to managing the farm, and to combine this with the cost of data collection. Gates and Kisiel (1974) evaluate the worth of additional data to a computer model of a groundwater basin. They focused on prediction errors caused by errors in basic data. They point out however, that their model is not able to indicate when further sampling is no longer justifiable. This is due to that they do not express sample worth in terms of precise economic benefit, but rather in terms of reduction of the expected error after sampling.

5.2. Early works on risk management frameworks

Sharefkin et al. (1984) developed a generalised cost-benefit analysis as a framework for evaluating the impacts, costs, and techniques for mitigating groundwater contamination. The purpose of the paper was to investigate economic analyses of groundwater contamination problems and economic comparison between alternative policies for managing groundwater contamination. They compare different techniques of surface water control, groundwater flow control, plume management, chemical immobilisation, and excavation and reburial. A damage valuation is done to compare the costs of the different alternatives. The damage valuation is based on a health-risk assessment of summing up the individual-chemical cancer induction risks, ignoring synergistic effects. The damage valuation is done in economical terms by multiplying the mortality risk range with the value of the mortality risk range. They list some recent values of life estimates varying from about US$ 10 million to US$ 57 000/life and apply a range of $\$10^5$-$10^6$. Results suggest that the potential damages and the cost of containment once contamination has occurred can be quite high and they conclude that prevention appears to be the best cure in these situations.

Marin et al. (1989) proposed a sequential decision framework for assessing the effects of waste sites on groundwater. The methodology was incorporated into an advisory computer system for North Carolina groundwater quality modelling and management needs. The system uses Monte Carlo simulation techniques with a deterministic model incorporated into a sequential Bayesian risk methodology. It is stated to provide a flexible decision format within which to consider permitting and monitoring of hazardous waste sites under imperfect information.

The criterion used for ranking alternate permit strategies is based on the corresponding expected net benefits. The net benefits are in principle calculated by subtracting the cost of supplying control level L_i from the benefits of the avoided hazard costs. Supply costs are by far not as difficult to estimate as are the avoided hazard costs since the latter addresses human health damage costs. Marin et al. (1989) use a proxy objective by using the appropriate level of protection as the one which meets the standards set by regulatory agencies at minimum cost. The advisory system of Marin et al. (1989) is designed to represent the permitting process as a sequential set of logical decision points. It involves a macro program that sequences the execution of various components of the system: data handling and analysis, model selection and use, risk/error analysis, sampling design, and permitting decision. The decision is obtained by proceeding through an increasingly refined set of criteria. In a second paper (Medina et al. 1989) they present the modifications used to adapt a deterministic numerical transport model for Monte Carlo analysis in the sequential algorithm.

5.3. Risk-cost-benefit analysis

The most extensive work on risk-cost-benefit analysis applied in hydrogeological design is that by Freeze et al. (1990). It was presented in four-part article series in Ground Water during the years 1990-1992. The first part emphasises the outline of the decision framework, the second and third parts (Massmann et al., 1991; Sperling et al., 1992) gives examples of applications to groundwater contamination and groundwater control systems respectively. The third part (Freeze et al., 1992) investigates the use of data worth in the development of site investigation strategies. Figure 5.1 gives the outline of the framework, with its different parts.

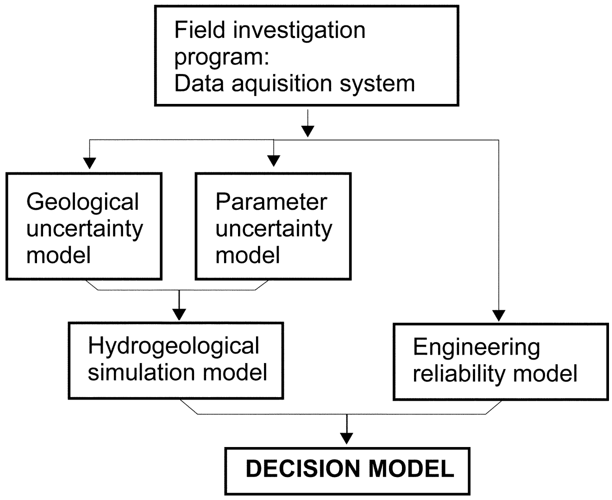


Figure 5.1. The outline of the proposed decision framework by Freeze et al. (1990).

The decision model allows for comparison of alternatives. The engineering reliability model is used to represent the expected performance of engineered components of the system. The hydrogeological simulation model is used to represent the expected performance of the hydrogeological parts of the system. The simulation must be stochastic to account for uncertainties, described by a geological uncertainty model and a parameter uncertainty model. The field investigation program determines the form of the uncertainty models, where the concept of data worth is included. For a detailed description of the framework, the reader is directed to these papers. Below is only a short summary of the decision model.

The decision model is based on a risk-cost-benefit analysis of each possible decision alternative. The risk is a probabilistic cost, whereas the investment costs and the benefits are viewed as being completely known. The alternative that has the lowest total cost should be chosen. The total cost is given by an objective function. The objective function (Φ) for each alternative j = 1…N is the net present value of the expected stream of benefits (B), costs (C) and risks (R) over an engineering time horizon (T), and discounted at the market interest rate (i):

$$\Phi_j = \sum_{t=0}^{T} \frac{1}{(1+i)^t} \left[B_j(t) - C_j(t) - R_j(t) \right]$$

where Φ_j is the objective function for alternative j [$]; $B_j(t)$ are the benefits of alternative j in year t [$]; $C_j(t)$ are the costs of alternative j in year t [$]; $R_j(t)$ are the risks of alternative j in year t; T is the time horizon [years]; and i is the discount rate [decimal fraction].

The risk is an engineering risk to the owner-operator as opposed to a health risk to receptors in a regulatory health-risk characterisation. The risks R(t) are defined as the expected costs associated with the probability of failure:

$$R(t) = P_f(t) C_f(t) \gamma(C_f)$$

where $P_f(t)$ is the probability of failure in year t [decimal fraction], $C_f(t)$ are the costs associated with failure in year t [$], and $\gamma(C_f)$ is a normalised utility function [decimal fraction, $\gamma \geq 1$] to be able to account for risk-averseness in the objective function. P_f is calculated from the uncertainty models in the framework.

The optimal risk is considered to be the risk that gives the lowest total cost. Sometimes this theoretical optimal risk is not accepted i.e. regulatory agencies. Figure 5.2 describes the concept of optimal risk as opposed to acceptable risk. Figure 5.3 is a summary that gives an overview f the different components that must be considered when designing a decision model.

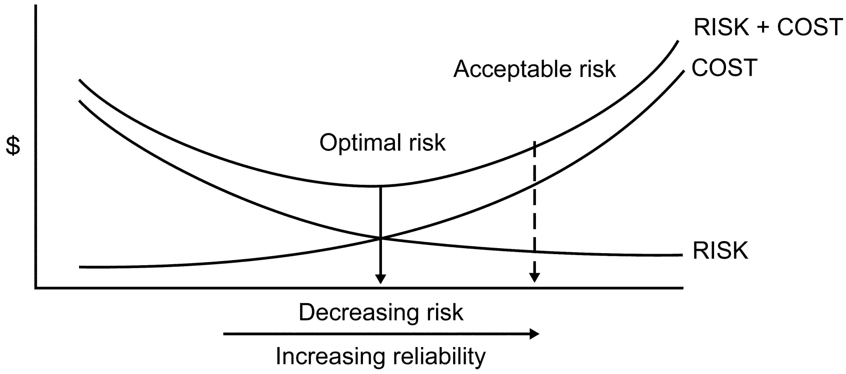


Figure 5.2. The optimal risk is where the total cost is minimised. The optimal risk may not be acceptable however. The acceptable risk can be at a higher total cost. After Freeze et al. (1990).

In principal, Freeze et al. (1990) summarised and completed much of what had previously been done in the field of decision analysis applied at hydrological and hydrogeological design. James et al. (1996b) states that is an important formalised tool for helping site managers to efficient allocation of resources under conditions of uncertainty, complex sites and diverse stakeholders. Two decisions in particular are addressed with risk-cost-benefit analysis in hydrogeology: (1) determination of the lowest-total-cost remedial action alternative from a suite of acceptable alternatives, and (2) estimation of the cost-effectiveness of collecting additional information to reduce uncertainty.

Rosén and LeGrand (1997) present a preliminary guidance framework in accordance with Freeze et al. (1990), for monetary risk assessments of groundwater contamination at early stages, prior to any new measurements or actions. Two objectives of the framework is put forward: 1) to provide an assessment framework which optimises use of professional judgement for studies where data are limited and 2) to give synergistic interpretative values that complement field measurements and that can be used as prior estimates in more detailed studies. The development of conceptual models, based on sound hydrogeological reasoning by experienced hydrogeologists is pointed out as a key issue to arrive at useful risk assessment at early stages. Four basic principles are applied within the guidance framework presented: 1) Bayesian statistics, 2) the concept of exceeding critical compliance levels, 3) hypothesis testing, and 4) a dual-oppositional site approach.

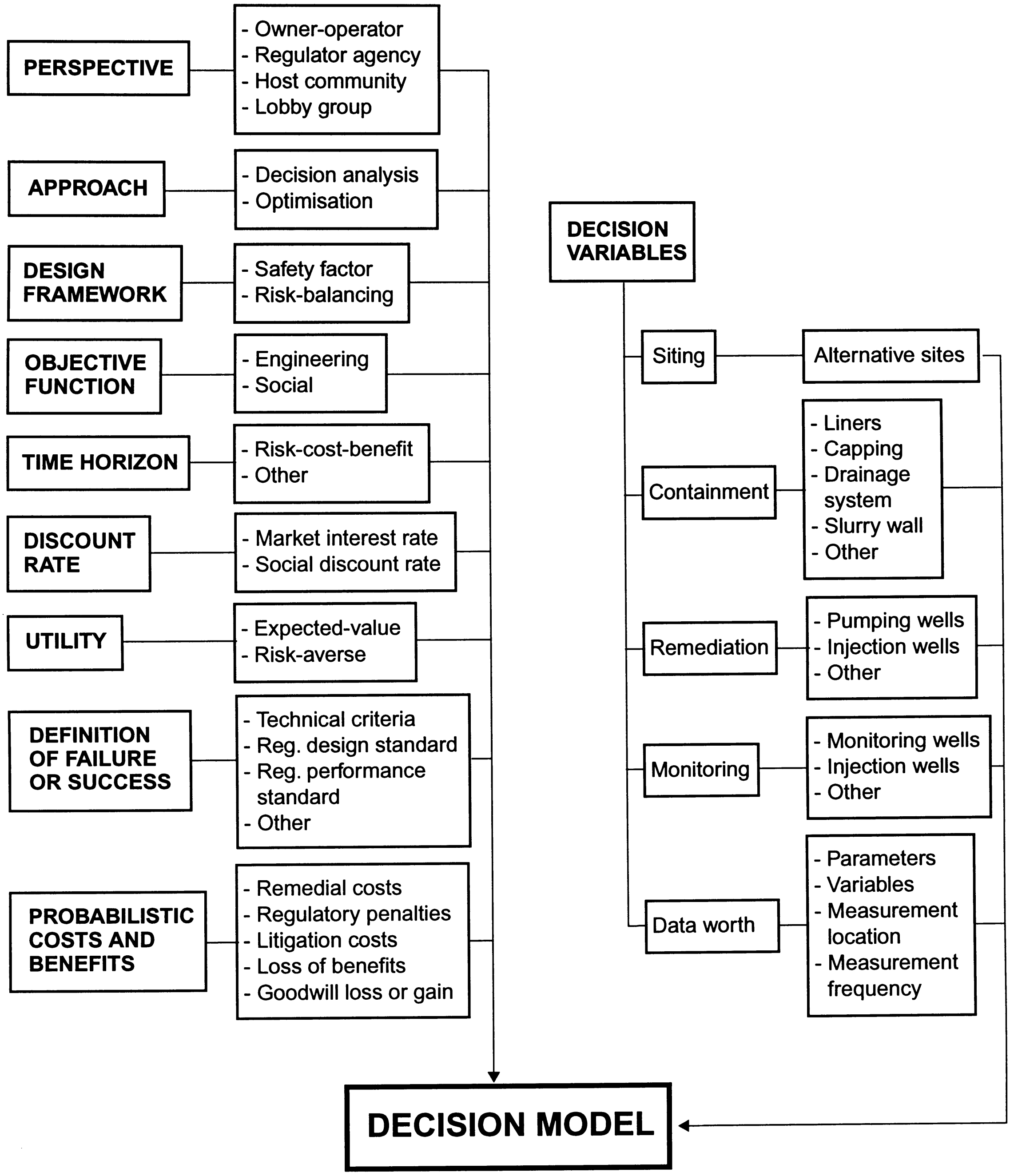


Figure 5.3: An overview f the different components that must be considered when designing a decision model. After Freeze et al. (1990).

5.3.1. Examples of case studies with the application of risk-cost-benefit analysis

Massmann and Freeze (1987a, b) used an earlier version of the described framework to investigate the interaction between risk-based engineering design and regulatory policy applied to groundwater contamination from waste management sites. They did this by applying a risk-cost-benefit analysis from an owner/operator perspective and indirect analyse the impacts of different regulations. The probability of failure in the study is calculated by means of two probabilities: 1) the probability of breach of containment (reliability theory) and 2) probabilistic contaminant travel times (by numerical simulations of advective travel time in a stochastic medium). Massmann and Freeze (1987b) include a case history with the principal motive to illustrate that the relatively large amount of data required for the analysis presented in this study can be obtained for a fairly typical application. The risk term in the owner/operator's objective function was found to be of relative unimportance. If this would be found to be a common economic feature for a more general suite of landfill sites, it would lend credence to their conclusion that regulatory agencies must use design standards and siting standards rather than performance standards and penalties to ensure groundwater quality.

An example given by James et al. (1996b) is demonstrated to a real remediation problem at Oak Ridge National Laboratory in Tennessee where radioactive waste was buried in earthen trenches, which is now leaching. The options considered are whether to provide an interim containment immediately, while waiting for a permanent remediation method or whether monitoring alone is acceptable. James et al. (1996b) present the framework as a three-step process: 1) prior analysis, 2) data-worth analysis, and eventually 3) posterior analysis based on new information if this was found to be cost-efficient. Cost-effectiveness of new data is defined as when the new data will change the outcome of the prior risk-cost-benefit analysis. A maximum justifiable exploration budget is estimated by using the expected value of perfect information (EVPI). EVPI is explained by the use of economic regret. The analysis is simplified in that probabilities are estimated instead of calculated. By reasoning it is shown that rather robust decisions can be made with sparse data.

Jardine et al. (1996) used a decision analysis approach as described by Freeze et al. (1990) to the design of a performance monitoring network at a waste management facility overlying fractured bedrock. The objective was to detect contaminants before they reach a regulatory compliance boundary and the features considered in the monitoring network include the number of wells and their locations, the discrete monitoring zone in each borehole and how often to take water samples. They used a discrete fracture model to simulate flow in a two-dimensional network of planar fractures. Selroos (1997a, b) is inspired by the hydrogeological decision-analysis framework given by Freeze et al. (1990) for developing a stochastic-analytical framework for safety assessment of waste repositories. Nonreactive transport and various linear mass transfer processes are accounted for.

Russell and Rabideau (2000) use Decision Analysis (DA) as presented by Freeze et al. (1990) to evaluate 27 alternatives of pump-and-treat design. They did a thorough analysis of a large number of factors that may influence a decision. For each of the 27 alternatives they varied aquifer heterogeneity, the way of defining the system failure, the definition of clean-up standards and different levels of failure costs. Additionally, the complexity of the simulation model was varied in two different ways. Generally, heterogeneity, failure definition, and failure cost influenced the outcome of the DA. The use of different complex simulation models seemed to generate similar outcomes. This is however, not thoroughly discussed in the paper.

Variations of P_f

Lepage et al. (1999) present a case study at a large municipal landfill in Montreal. It follows Freeze et al. (1990) in principle but ignores to quantify uncertainties of the geology, hydrogeological parameters and the reliability of the technical system. The decision analysis is done from an owner/operator perspective using a market interest rate. The performance criterion to meet is the containment of leachate within the property boundaries, making a flow model suitable, as the concentration is not an issue. Four different alternatives are evaluated, all based on the idea of hydraulic containment. The probability of failure is not quantified as a probability since the uncertainties are not quantified. Instead a weighting factor is used to calculate the risk. The failure weighting factor (β) is defined as the ratio of the number of streamlines leaving the site for the tested operating conditions, divided by the number of streamlines leaving for the calibrated model without any new leachate hydraulic control measure.

Barnes and McWhorter (2000a) propose a risk-cost approach for the design of soil vapour extraction (SVE) system that takes into account uncertainties in the soil properties. The framework uses three of the six components first proposed by Freeze et al. (1990): 1) a hydrogeological simulation model, 2) a parameter uncertainty model, and 3) a decision model. A two-dimensional soil-gas flow and vapour transport numerical model, VapourT was used. The only parameter that was assumed uncertain was the intrinsic permeability, which was inserted in the governing equation by means of a Monte Carlo simulation. The authors point out that other parameters may as well be considered as uncertain (for example porosity and fraction of organic carbon). A risk-neutral decision-maker is assumed, the time horizon was set to 10 years, and the costs and risks time-dependent. Cost of failure was estimated as the cost of excavating and disposing of the entire volume of originally contaminated soil. Probability of failure was calculated as the ratio of the number of realisations that fail to meet the cleanup goal to the number of realisations used in the Monte Carlo simulation, similar to the idea of Lepage et al. (1999).

5.3.2. Including uncertainties in C_f

Wladis et al. (1999) uses the decision model formulated by Freeze et al. (1990) to perform a risk-based decision analysis of atmospheric emission alternatives to reduce groundwater degradation on the European scale. The main objectives of their work are to provide a conceptual framework for 1) analysing economic implications of the hydrogeologic uncertainties in decision analyses of the different alternatives, and 2) studying the impact of different approaches to value groundwater degradation. The SMART2 model (Kros et al., 1995) was used to predict concentrations of aluminium and nitrate in groundwater due to input of acidifying components from the atmosphere. Two atmospheric emission control scenarios for NO_x, SO_2, and NH_3 were studied with respect to their effects of reducing nitrate and aluminium contamination in natural land in the Netherlands. Probability density functions (PDF) were assigned to the model inputs and Monte Carlo simulation yielded PDFs of the block concentrations and contamination areas. Benefits and implementation costs were estimated.

Cost of failure was estimated both by direct and indirect methods. Wladis et al. (1999) estimated a minimum value of groundwater resources and applied a range of economic values to the resource to study the sensitivity of decision analysis to the valuation of groundwater. In total eight different values of the cost of failure were prepared. The study was done from a societal perspective, using a zero discount rate. The results indicate that the decision on the optimum alternative is dependent on the cost of failure and the acceptable level of economic uncertainty being equal to the variability of the risk (Wladis et al., 1999). They summarise it as; the larger the reduction of emissions, the less the economic uncertainty and the higher the value of the groundwater resource, the larger the economic uncertainty.

5.3.3. Including Regret

Dakins et al. (1994) uses a similar approach as Freeze's group does for deciding how many cubic meters of sediments that must be dredged in New Bedford Harbor, Massachusetts in order to reach the environmental goal of a maximum PCB-concentration in winter flounders. They use two failure criteria to calculate the expected loss of a certain management decision. 1) Over-remediation; the cost of failure is the amount dredged unnecessarily. 2) Under-remediation; the cost of additional dredging plus the cost of delaying the dredging of the harbour, thus delaying the fishing industry. They estimate the dredge cost to $1000/m^2. The cost of under-remediation is estimated to $50 million. A_d is the area dredged given the decision and A_c is the correct (but unknown) area necessary to dredge just to meet the PCB-criterion. Then the loss function (L) can be written as:

$$L(A_d|A_c) = \$1{,}000A_d \qquad A_d \geq A_c$$
$$L(A_d|A_c) = \$1{,}000A_d + \$50{,}000{,}000 \qquad A_d < A_c$$

Unsurprisingly, we don't know the correct area and are forced to calculate the *expected* loss instead. This is done using Monte Carlo simulation, computing the average loss for a fixed A_d taken over the probability space of A_c:

$$E[L(A_d)] = \frac{1}{N}\sum_{i=1}^{N} L(A_d|A_{c,i})$$

$A_{c,i}$ is the correct area to be dredged based on the *i*th iteration of the Monte Carlo simulation. The optimal decision is identified as the alternative with the minimal expected loss of a series of values of A_d.

Additionally they compute the expected value of including uncertainty (EVIU) in their analysis as the difference between the expected loss of the optimal management decision based on a deterministic analysis and the expected loss of the optimal management decision based on the uncertainty analysis. The expected value of perfect information (EVPI) is calculated. The remediation decision done is always A_d=A_c in this case:

$$E[L(\text{perfect information})] = \frac{1}{N}\sum_{i=1}^{N} (\$1{,}000A_{c,i})$$

The uncertainty analysis was done considering inputs and parameters in the model of PCB-uptake in winter flounders, whereas the model structure is not included. Monte Carlo simulation using the Latin Hypercube sampling technique is used.

Angulo and Tang (1999) uses two failure criteria similar to Dakins et al. (1994) to design the most optimal groundwater detection monitoring system under uncertainty. They use Monte Carlo simulation for the calculations. The *expected* total cost is a function of construction and monitoring costs (C_c) and cost of remediation (C_v) given the monitoring system:

$$E(\text{Total cost}) = C_c + P_d C_v E(v|d) + P_f C_v E(v|f)$$

where P_d is the probability of detection, P_f is the probability of failure of the system to detect contaminants, E(v | d) and E(v | f) are the expected plume volume given detection and failure to detect a plume, respectively. The construction and monitoring cost depends on the number of wells in the system. They aim to maximise the probability of detection, to minimise the contaminated volume, and to minimise the total cost.

5.3.4. Alternative decision criteria

Paleologos and Lerche (1999) points at the need for using more than one decision criteria in environmental projects, exemplified with the transport and burial of hazardous and radioactive wastes. The decision criterion often used is to maximise the expected monetary value (MEMV). The authors conclude that the sole use of this criterion may lead to erroneous decisions in the presence of uncertainty. Instead the authors propose to use additional statistical measures such as for example the standard error and volatility (a measure of the uncertainty of the expected value) to get insight into the decision process. The reason is that MEMV fails to differentiate between the consequences of limited and catastrophic failures. Including high-cost and low-probability events in a decision analysis can have different effects such as: (I) Reversing the expected return from a positive to a negative value for a range of contract awards of a project. (II) Significantly increasing the standard error and volatility. (III) Substantially reducing the probability of success. An unjustified inclusion of catastrophic scenarios can alter the perspective of a project and guide a corporation away from a possibly profitable investment.

5.3.5. Data worth analysis using different investigation strategies

The concept of data worth analysis is based on Bayesain updating of prior estimates. The idea is to make the updating before measurements, that is before you know the actual test results. Thus you need to be able to predict how much the proposed investigation program will reduce uncertainty. The proposed investigation programs are commonly analysed by means of search theory or indicator kriging.

Freeze et al. (1992) includes data worth assessment in their proposed decision analytical framework. Figure 5.4 should thus be seen in relation to previous figure 5.3. In the decision model goes the prior estimates. A data worth analysis is done using preposterior estimates. Preposterior estimates are updated by Bayes' theorem using the expected result of a sample. If the new (unknown before actual measurement) data increases the expected value of the objective function more than the cost of obtaining that data, then the data are worthwhile. The concept of regret is commonly included in data worth analysis (see also expected opportunity loss, part 5.4.3). Thus, if the difference between the regret prior to new data and the regret (pre-) posterior to new data is larger than the cost of that data, then it is worthwhile.

Freeze et al. (1992) described the concept of data worth for detecting aquitard discontinuities using search theory. This was further developed in James and Freeze (1993) where they used indicator kriging to estimate the reduction of uncertainty of the presence or absence of a hydrogeological window in the aquitard.

James and Gorelick (1994) propose a stepwise optimising framework for the use of data worth analysis in the application of delineating and remediating a contaminant plume. Uncertainties lay within uncertain source location and loading time and aquifer heterogeneity. The objective is to minimise the total cost of sampling and remediation. Monte Carlo simulation was used to generate a number of equally likely plumes. A number of these plumes were randomly drawn and for each of these the optimum number and location of samples were estimated. The optimal number and location of sample points is the average of these.

Figure 5.5 shows the proposed structure, which also applies to the work of Freeze et al. (1990).

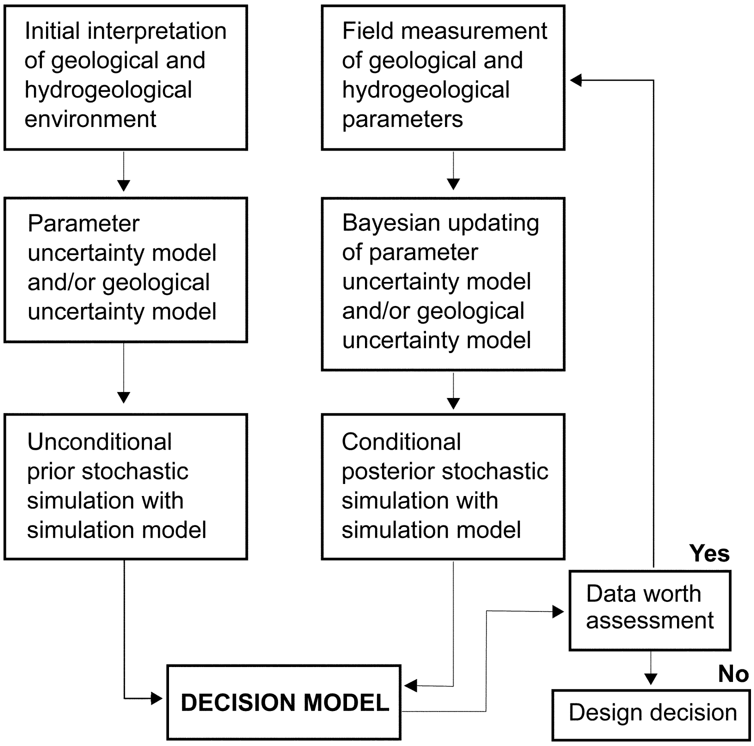


Figure 5.4. Overview of decision framework with data worth analysis included. After Freeze et al. (1990).

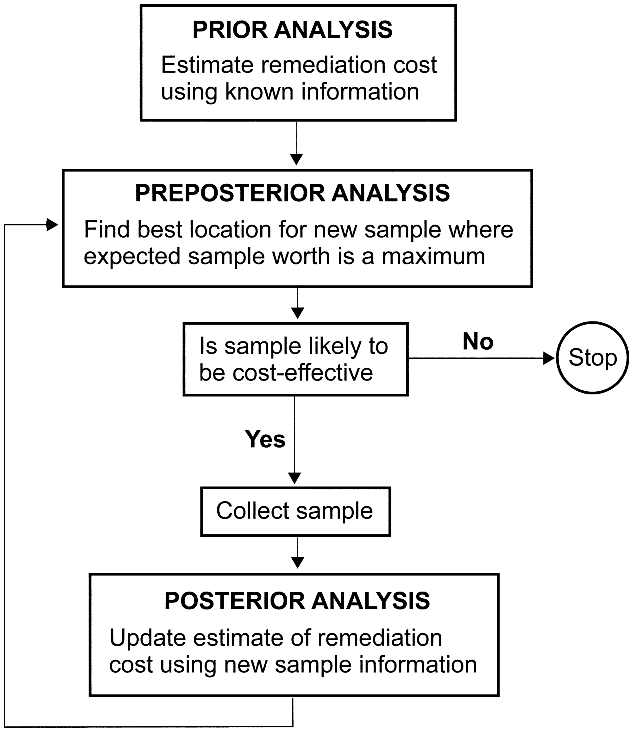


Figure 5.5. Outline of steps included in data worth analysis. After Freeze et al. (1990).

Dakins et al. (1996) used a similar approach on the problem of risk-based remediation of PCB contaminated sediments. They call it Bayesian Monte Carlo analysis and it is used to simulate outcomes of future data collection programs. This is in principle what is described above. They calculate the expected value of sample information (EVSI) for each propsed sampling program and compares this with the expected value of perfect information (EVPI). EVPI acts as a upper limit to how large (expensive) the sampling program can be. They conslude that EVSI is most sensitive to the unit cost of remediation but rather insensitive to the failure cost of under-remediation (see also Dakins et al., 1994, previously described in part 5.4.3).

James et al. (1996a) uses the concept of regret to analyse whether new data is of any value, that is if it has the potential to change the decision outcome. They determine parameters where the uncertainty has the largest effect on the outcome by a ranking method using regional sensitivity analysis (RSA). They note that there can be four possible outcomes of a sampling program: 1) conditions for failure are correctly indicated, 2) conditions for failure are falsely indicated, 3) non-failure conditions are correctly indicated, and 4) non-failure conditions are falsely indicated. For simplicity they assume that any sampling program will not give any false indication of failure.

5.4. Social decision theory

Kruber and Schoene (1998) propose that remediation decisions with different interested parties can greatly benefit from decision analysis. Figure 5.6 shows the main steps of the proposed decision analytic procedure. The decision context outlines what objectives and restrictions (financial limitation, regulatory restrictions etc.) are imposed on the decision at hand. Three remediation companies was contacted to do the design and quantification of decision alternatives. The authors added a fourth technology to enlarge the scope of the decision. Uncertainties are represented by probability density functions. The result is a matrix that for each criterion and each remediation alternative contains a probability distribution.

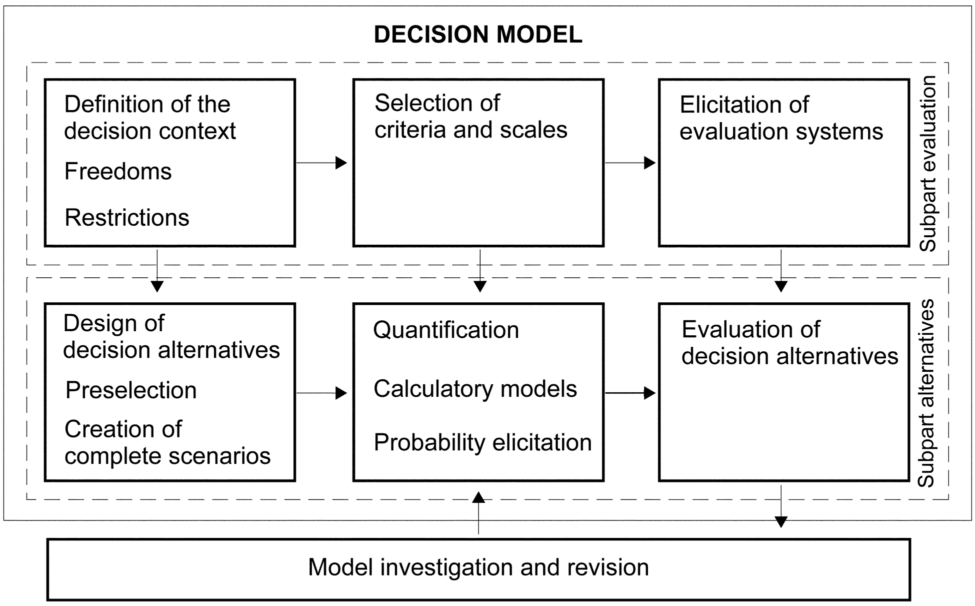


Figure 5.6. The main steps of the proposed decision analytic procedure by Kruber and Schoene (1998).

The preferences of each decision-maker are represented as a formal, mathematical model in the defined evaluation system. The evaluation system assigns to every possible result on the criteria a real number, the total utility. Two building blocks are part of an evaluation system: 1) non-linear functions over the criteria, called utility functions, which express the risk attitude of the decision-maker with respect to one criterion and 2) an aggregation method that collapses all criteria into one indicator. Kruber and Schoene (1998) note that an evaluation system can be constructed for each decision-maker, which can work as a starting point for negotiations between the different parties. Utility functions are based on the concept of trade-off between criteria.

The expected value of the total utility is calculated for each alternative once the evaluation systems and the probability distributions are determined, giving a complete ranking of the decision alternatives. The ranking of the decision alternatives is just the first step of the solution of a decision problem according to Kruber and Schoene (1998). The complete mathematical decision model gives the possibility to investigate for example which facts were most important for the observed ranking or the stability of the ranking with respect to variations of the input data. Sensitivity analyses can be done to identify the most crucial uncertainties. The authors also suggest a project attendant to assist the people involved in remediation decisions in building the decision model.

Renn (1999) proposes a model for an analytic-deliberative process in risk management, named cooperative discourse. The model consists of three steps: elicitation of values and criteria by the stakeholder groups; provision of performance profiles for each policy option by experts; and evaluation and design of policies by randomly selected citizens. The model aims at citizen participation and public involvement in risk management because risk perception is usually different between the public and experts/policy makers. The different actors in the cooperative discourse model are: stakeholder groups, experts, citizens, sponsors, and a research team. The author means that the mere desire to initiate a two-way-communication process and the willingness to listen to public concerns is not enough. A structure that assures the integration of technical expertise, regulatory requirements, and public values is needed.

6. DISCUSSION

This literature review has presented a broad spectrum of issues. The reason for this is that risk analysis is not an isolated field, but a field that interacts with the technical, economical, political and ethical arenas. Numerous books, papers and reports have been published since the 70'ies concerning these issues. The main aim of this report is to provide a scientific basis for a doctoral project on risk-based decision analysis, and to be able to draw some conclusions regarding the continuation of this project.

The criterion to maximise the expected utility is by all means the decision criterion most widely used, often expressed as risk-cost-benefit analysis. Risk-cost-benefit analysis has also had a special focus in this report, since we have been interested to investigate the use of a framework based on this theory. Even though cost-benefit analysis is a well-known tool to many companies the use of risk-cost-benefit analysis is somewhat controversial.

A discussion on risk-cost-benefit analysis can take place at different levels. I will discuss it primarily on two levels: 1) if it is appropriate to use and, 2) if it is considered to be appropriate, the advantages and practical limitations of using such a framework. Finally, some conclusions regarding future work will be discussed.

The first discussion is of a moral-philosophical or ethical character but nonetheless important. Most of the discussion can easily be directed towards the issue of valuing human health and the environment in monetary terms. Some of the first frameworks considered the monetary valuation of a statistical life, but this seems to be less interesting recently. Instead we have environmental standards that often are based on the assumption that an incremental risk over a lifetime of 10^{-6} for developing cancer is acceptable. Facing the fact that we can not get a society without any risks at all, this seems as a less controversial method than monetary valuation of lives. However, the uncertainty of the outcome of a risk assessment needs to be made visible.

A common argument for the valuation of environmental goods is that it can be fitted into traditional decision-making based on cost-benefit analysis. The conclusion often drawn is that if it is not valued it will not be considered at all. Those who argue against this mean that it is immoral to value the environment at all and that this should be considered separately from the economy. There is, however, an increasing trend in the society towards explicit valuation of our natural resources. The reason may be ethical; to explicitly account for environmental good but it may as well be a compromise between a bad (not considering environmental effects at all) and a better but non-optimal alternative.

Risk-cost-benefit analysis defines consequences in monetary terms and can be subject to these critiques. However, reality is not black or white. Whereas in some cases risk-cost-benefit analysis can be considered to be out of the question it may well serve its purposes in other situations. The degree of appropriateness depends on the art of the consequences. If consequences are e.g. irreversible or of very large scale a risk-averse behaviour is often well motivated. The degree of appropriateness also depends on the art of the prevailing uncertainties, if they are very large or small, or even unknown.

Recognising these difficulties, risk-cost-benefit analysis should be considered a tool to structure complex problems, not a tool to provide an objective true best answer. This is most obvious when discussing different decision perspectives. A societal decision-maker will have to, at the same site, consider a different decision horizon, regarding both consequences and time horizon, as compared to a private decision-maker. It is important to keep in mind the fact that what does not go into the analysis will not be considered and that the decision-maker are responsible for what goes into the analysis.

The discussion on the ethical aspects of using risk-cost-benefit analysis could be much more extensive. Basically, this is a discussion of whether utilitarianism fails to consider important aspects such as justice, equity and freedom. However, this is somewhat out of the scope of this report.

When it may be considered to be appropriate to use risk-cost-benefit analysis in a project, there are some topics of a more practical character which are of interest.

Obviously, the inclusion of uncertainties in the analysis is of primary importance. When data is insufficient to make statistical estimations of these uncertainties, we are forced to make subjective estimations of those uncertainties instead. However, for projects related to contaminated sites it may well be a large *advantage* of being able to include subjective information. Thus, expert knowledge is something that should be aimed at making the best use of. But as many authors concluded, estimations are often far too overconfident which is an aspect that needs attention. Bayesian statistics makes use of subjective information. For the purpose of data worth analysis included in the framework, Bayesian statistics makes it possible to update prior data.

The uncertainties needs to be included in calculations on contaminant transport and groundwater flow, in the decision model and in the consequence costs. The idea of risk-cost-benefit analysis however, is not primarily the use of specific software, but rather to apply a certain way of thinking about uncertainties and weighing these uncertainties against costs. Software, modelling codes and similar are important tools to do this. There are numerous modelling tools available and the choice of model is an important aspect in risk-cost-benefit analysis. The model must be able to incorporate uncertainties and it should be chosen in relation to the decision to be taken. If the consequences may be very large, a higher degree in complexity of the model is well motivated. On the other hand simplified analytical solutions on contaminant transport may be sufficient for decision problems with less impact or at an early stage of a decision analysis.

Consequence costs can be difficult to estimate. It is important to carefully describe in words what the consequences are and as far possible quantify them for the purpose of risk-cost-benefit analysis. Some authors use different kind of valuation methods to estimate environmental costs, others choose to consider only consequences that are easily related to monetary costs. If risk-cost-benefit analysis for remediation of contaminated sites is to become a method that inspires confidence in a societal perspective, valuation of environmental consequences are needed and they are needed to be done by professionals. Discussions and estimations of the relative size of the consequence costs may also be useful for an overall picture of the problem at hand.

But let us focus on some possible advantages of using a risk-cost-benefit approach when making decisions at contaminated sites. Generally, it is in the initial planning phase of (any) projects that it is possible to influence the costs that will occur later during the execution phase in the project. In this planning phase, when limited data is available, it is thus of importance to be able to make the best use of data. The effort spent in the planning phase should also be in balance with the extent of the consequences and the decision to be made.

The project should be decision-driven. Risk-cost-benefit analysis can be a way of structuring problems consequently from the early stages of a project to make optimal use of data. It is in the nature of risk-cost-benefit to focus on the decision to be taken, forcing the decision-maker to identify the goal in an early stage.

To structure problems and including uncertainties is not commonly used yet in Sweden in projects dealing with investigation and remediation of contaminated soil. Including uncertainties requires an understanding for how to interpret the results. Hence, the presentation of such analyses and results are in great need for being clear and understandable.

The presentation of material is closely connected to communication within the project but also communication with affected parties that are not directly involved in the project. An advantage with using a structured approach such as risk-cost-benefit analysis is that the data needed in the analysis have to be provided from different persons with different knowledge. Thus, it forces a communication between different parties throughout the project.

The advantages of using a risk-cost analytical approach can be summarised as:
1. Making uncertainties visible.
2. Structuring of complex decision situations.
3. Identification of the most cost-efficient decision alternative to the decision-maker.
4. A basis for communication.

A necessary part of the doctoral project is to analyse the gain of applying a decision theoretical view on common problems. E.g. Dakins et al. (1994) include EVIU (expected value of including uncertainty) in their analysis to investigate the benefit of including uncertainties. Of course, it would be nice if we could show an economical benefit of including uncertainties. In principal though, this can only be done when a number of projects have been carried out and evaluated since the benefit that may be calculated is the *expected* benefit. Important though, is to compare and judge what possible benefits may such an approach have in comparison with an approach not using such framework.

Worth mentioning is how uncertainties are treated today. The most common way of treating uncertainties is the use of safety factors. Often, a number of safety factors are added in different stages. The precautionary principle also tells us to be on the safe side. On the other hand, The Swedish EPA (Statens Naturvårdsverk, 1999b) gives advice to make use of a bad but probable case and not the worst case when making the risk classification of a contaminated site. With a risk-based approach, the idea is to make visible the uncertainties at hand and make decisions consciously of these uncertainties. In some cases it may be well motivated to have a risk-neutral approach. At other times, the consequences may be of such kind that a safe, or risk-averse, decision should be applied.

To summarise the future points of special interest for the project:

- To investigate the impact of too overconfident prior estimations of different variables and to understand how subjective information can be formally incorporated in risk analysis.

- To use the decision framework to evaluate and to understand the impact of choosing different decision criteria and different model complexity to different types of decision situations.

- With wide probability density functions and many variables the outcome of a decision analysis may have a very wide distribution. How can such wide distributions be interpreted and are they useful to decision-makers?

- To get experience with reasoning about environmental values. It will probably not be possible to get exact valuations of consequences, especially not when they are of more “existence value-character”, but how can incomplete valuations be used in a valuable way for decision analysis?

- To compare the difference of a decision analysis outcome using a private decision perspective and a societal decision perspective. It can not be expected that a decision analysis would get the same outcome for different decision perspectives. It is, however, interesting to analyse and discuss these differences in order to be able to motivate the use of such a framework both for a private as well as a societal decision-maker.

- To get practical experience of using risk-cost-benefit analysis and data worth analysis at contaminated sites. One of the main challenges is this practical aspect of applying a decision theoretical framework for evaluation of contaminated sites. How can this practically be incorporated in the early stages of projects?

7. REFERENCES

Abbaspour, K. C., Schulin, R., van Genuchten, M. Th. & E. Schläppi, 1998. Procedures for Uncertainty Analyses Applied to a Landfill Leachate Plume. Ground Water. Vol. 36, no. 6, pp. 874-883.

Alén, C., 1998. On probability in geotechnics: Random calculation models exemplified on slope stability analysis and ground-superstructure interaction. Volume 1. Doctoral thesis. Department of Geotechnical Engineering. Chalmers University of Technology. Göteborg. Sweden.

Al-Yousfi, A. B., Hannak, P. G., Strunk, Jr., J. F., W. V. Davies & S. I. Shah, 2000. Risk-Based Zoning Strategy for Soil Remediation at an Industrial Site. Journal of Soil Contamination. Vol. 9, No. 1, pp. 1-12.

American Chemical Society & Resources For the Future, 1998. Understanding Risk Analysis – A Short Guide for Health, Safety and Environmental Policy Making. Internet edition. (2000-12-19). http://www.rff.org/misc_docs/risk_book.htm.

American Society of Testing Materials (ASTM), 1995. Standard Guide for Developing Concenptual Site Models for Contaminated Sites. In Annual Book of ASTM Standards. Designation E 1689 – 95. USA.

Andersson, C., 1999. Groundwater Transport in Environmental Risk and Cost Analysis: the Role of Random Spatial Variability and Sorption Processes. Licentiate Thesis. Division of Water Resource Engineering. Department of Civil and Environmental Engineering. Royal Institute of Technology. Stockholm. Sweden.

Andersson, I. & T. Lindvall, 1995. Riskbedömning – Hälsa – Miljö. (In Swedish). Rapport 4409. Naturvårdsverket. Stockholm. Sweden.

Angulo, M. & W. H. Tang, 1999. Optimal Ground-Water Detection Monitoring System Design under Uncertainty. Journal of Geotechnical and Geoenvironmental Engineering. Vol. 125, no. 6, pp. 510-517.

Asante-Duah, D. K., 1998. Risk Assessment in Environmental Management. John Wiley & Sons Ltd. Chichester.

Balson, W. E., Welsh, J. L. & D. S. Wilson, 1992. Using Decision Analysis and Risk Analysis to Manage Utility Environmnetal Risk. Interfaces 22. No. 6, pp. 126-139.

Barnes, D. L, & D. B. McWhorter, 2000a. Design of Soil Vapor Extraction Systems Under Conditions of Uncertainty. Ground Water. Vol. 38, No. 2, pp. 209-217.

Barnes, D. L. & D. B. McWhorter, 2000b. Uncertainty in Predicting the Rate of Mass Removal Created by Soil Vapor Extraction Systems. Journal of Soil Contamination. Vol. 9, no. 1, pp. 13-29.

Batchelor, B., Valdés, J. & V. Araganth, 1998. Stochastic Risk Assessment of Sites Contaminated by Hazardous Wastes. Journal of Environmental Engineering. Vol. 124, no. 4, pp. 380-388.

Bell, D. E., 1982. Regret in Decision Making under Uncertainty. Operations Research. Vol. 30, No. 5, pp. 961-981.

Bethke, C. M. & P. V. Brady, 2000. How the K_d Approach Undermines Ground Water Cleanup. Ground Water. Vol. 38, no. 3, pp. 435-443.

Brent, R. J., 1996. Applied cost-benefit analysis. Edward Elgar Publishing Limited. UK.

Canadian Association of Physicians for the Environment (C.A.P.E.), 2000. http://www.cape.ca/Children/neuro.html. 2000-12-20.

Cairney, T., 1995. The Re-use of Contaminated Land. A Handbook of Risk Assessment. John Wiley & Sons Ltd. West Sussex.

Copty, N. K. & A. N. Findikakis, 2000. Quantitative Estimates of the Uncertainty in the Evaluation of Ground Water Remediation Schemes. Ground Water. Vol. 38, no. 1, pp. 29-37.

Covello, V. T. & M. W. Merkhofer, 1993. Risk assessment methods: Approaches for assessing health and environmental risks. Plenum Press. New York.

Dagan, G., 1997. Stochastic modelling of flow and transport: the broad perspective. In Subsurface Flow and Transport: A Stochastic Approach. Eds. Dagan, G. & S. P. Neuman. International Hydrology Series. Cambridge University Press. UK.

Dakins, M. E., Toll, J. E. & M. J. Small, 1994. Risk-Based Environmental Remediation: Decision Framework and Role of Uncertainty. Environmental Toxicology and Chemistry. Vol. 13, no. 12, pp. 1907-1915.

Dakins, M. E., Toll, J. E., Small, M. J. & K. P. Brand, 1996. Risk-Based Environmental Remediation: Bayesian Monte Carlo Analysis and the Expected Value of Sample Information. Risk Analysis. Vol. 16, no. 1, pp. 67-79.

Davies, J. C. (ed.), 1996. Comparing Environmental Risks: Tools for Setting Government Priorities. Resources for the Future. Washington DC.

Davis, D. R. & W. M. Dvoranchik, 1971. Evaluation of the Worth of Additional Data. Water Resources Bulletin. Vol. 7, no. 4, pp. 700-707.

Davis, D. R., Kisiel, C. C. & L. Duckstein, 1972. Bayesian Decision Thoery Applied to Design in Hydrology. Water Resources Research. Vol. 8, no. 1, pp. 33-41.

Dou, C., Woldt, W., Dahab, M. & I. Bogardi, 1997. Transient Ground-Water Flow Simulation Using a Fuzzy Set Approach. Ground Water. Vol. 35, no. 2, pp. 205-215.

Felter, S. P., Dourson, M. L. & J. Patterson, 1998. Assessing Risks to Human Health from Chemicals in the Environment. In Handbook of Environmental Risk Assessment and Management. Ed. P. Calow. Chapter 2, pp. 9-23. Blackwell Science. Great Britain.

Ferguson, C., Darmendrail, D., Freier, K., Jensen, B. K., Jensen, J., Kasamas, H., Urzelai, A. & J. Vegter (eds.), 1998. Risk Assessment for Contaminated Sites in Europe: Volume 1. Scientific Basis. LQM Press. Nottingham.

Freeze, R. A., & S. M. Gorelick, 1999.Convergence of Stochastic Optimization and Decision Analysis in the Engineering Design of Aquifer Remediation. Ground Water. Vol. 37, No. 6, pp. 934-954.

Freeze, R. A., James, B. Massmann, J., Sperling, T. & L. Smith, 1992. Hydrogeological Decision Analysis: 4. The Concept of Data Worth and Its Use in the Development of Site Investigation Strategies. Ground Water. Vol. 30, No. 4, pp. 574-588.

Freeze, R. A., Massmann, J., Smith, L., Sperling, T., James, B., 1990. Hydrogeological Decision Analysis: 1. A Framework. Ground Water. Vol. 28, No. 5, pp. 738-766.

Freeze R. A., McWhorter, D. B., 1997. A Framework for Assessing Risk Reduction Due to DNAPL Mass Removal from Low-Permeability Soils. Ground Water. Vol. 35, No. 1, pp. 111-123.

Gates, J. S. & C. C. Kisiel, 1974. Worth of Additional Data to a Digital Computer Model of a Groundwater Basin. Vol. 10, no. 5, pp. 1031-1038.

Gorelick, S. M., 1997. Incorporating uncertainty into aquifer management models. In Subsurface Flow and Transport: A Stochastic Approach. Eds. Dagan, G. & S. P. Neuman. International Hydrology Series. Cambridge University Press. UK.

Gren, I-M., Söderqvist, T., Wulff, F., 1997. Nutrient reductions to the Baltic Sea: Ecology, costs and benefits. Journal of Environmental Management. No. 51, pp. 123-143.

Grosser, P. W. & A. S. Goodman, 1985. Determination of groundwater sampling frequencies through Bayesian decision theory. Civil Engineering Systems. Vol. 2, no. 4, pp. 186-194.

Haimes, Y. Y. & W. A. Hall, 1974. Multiobjectives in Water Resource Systems Analysis: The Surrogate Worth Trade Off Method. Water Resources Research. Vol. 10, no. 4, pp. 615-624.

Hamed, M. M. & P. B. Bedient, 1997. On the Performance of Computational Methods for the Assessment of Risk from Ground-Water Contamination. Ground Water. Vol. 35, no. 4, pp. 638-646.

Hamed, M. M., 1999. Probabilistic Sensitivity Analysis of Public Health Risk Assessment from Contaminated Soil. Journal of Soil Contamination. Vol. 8, no 3, pp. 285-306.

Hamed, M. M., 2000. Impact of Random Variables Probability Distribution on Public Health risk Assessment from Contaminated Soil. Journal of Soil Contamination. Vol. 9, no. 2, pp. 99-117.

Hamilton, G., 1996. Risk management 2000. (In Swedish). Studentlitteratur. Lund.

Hammitt, J. K., 1995. Can More Information Increase Uncertainty? Chance. Vol. 8, no. 3, pp. 15-17, 36.

Hammitt, J. K., 2000. Valuing Mortality Risk: Theory and Practice. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1396-1400.

Hammitt, J. K. & A. I. Shlyakhter, 1999. The Expected Value of Information and the Probability of Surprise. Risk Analysis. Vol. 19, no. 1, pp 135-152.

Hanley, N., Spash, C. L., 1995. Cost-benefit analysis and the environment. Second reprint. Edward Elgar Publishing Limited. UK

Hansson, S-O., 1989. Dimensions of Risk. Risk Analysis. Vol. 9, no. 1, pp. 107-112.

Hansson, S-O., 1990. An Overview of Decision Theory. Statens Kärnbränsle Nämnd (National Board for Spent Nuclear Fuel). SKN Report 41, 1991. Stockholm.

Hartlén, J., A-M. Fällman, P-E. Back, C. Jones, 1999. Principles for risk assessment of secondary materials in civil engineering work: Survey. AFR-Report 250. AFN, Naturvårsverket. Stockholm.

Hetrick, D. M. & A. Pandey, 1999. A Methodology for Establishing Cleanup Objectives in the Unsaturated Soil Zone Using Sensitivity and Uncertainty Analysis for Chemical Fate and Transport. Journal of Soil Contamination. Vol. 5, no. 8, pp. 559-576.

Hoffman, F. O. & J. S. Hammonds, 1994. Propagation of Uncertainty in Risk Assessments: The Need to Distinguish Between Uncertainty Due to Lack of Knowledge and Uncertainty Due to Variability.

Hohn, M. E., 1999. Geostatistics and Petroleum Geology. Second edition. Kluwer Academic Publishers. Dordrecht. The Netherlands.

James, B. R. & R. A. Freeze, 1993. The Worth of Data in Predicting Aquitard Continuity in Hydrogeological Design. Water Resources Research. Vol. 29, no. 7, pp. 2049-2065.

James, B. R. & S. M. Gorelick, 1994. When enough is enough: The worth of monitoring data in aquifer remediation design. Water Resources Research. Vol. 30, no. 12, pp. 3499-3513.

James, B. R., Gwo, J-P. & L. Toran, 1996a. Risk-Cost Decision Framework for Aquifer Remediation Design. Journal of Water Resources Planning and Management. Vol. November/December, pp. 414-420.

James, B. R., Huff, D. D., Trabalka, J. R., Ketelle, R. H. & C. T. Rightmire, 1996b. Allocation of Environmental Remediation Funds Using Economic Risk-Cost-Benefit Analysis; A Case Study. Groundwater Monitoring and Remediation. Vol. 16, no. 4, pp. 95-105.

Jardine, K., Smith, L. & T. Clemo, 1996. Monitoring Networks in Fractured Rocks: A Decision Analysis Approach. Ground Water. Vol. 34, no. 3, pp. 504-518.

Jettmar, R. U. & G. K. Young, 1975. Hydrologic Estimation and Economic Regret. Water Resources Research. Vol. 11, no. 5, pp. 548-656.

Johannesson, M., 1998. Risk Management Under Uncertainty: Staretgies for protecting health and the environment. Dissertation. Natural Resources Management. Department of Systems Ecology. Stockholm University. Stockholm.

Kaunas, J. R. & Y. Y. Haimes, 1985. Risk Management of Groundwater Contamination in a Multiobjective Framework. Water Resources Research. Vol. 21, No. 11, pp 1721-1730.

Keeney, R. L., 1982. Decision Analysis. Operations Research. Vol. 30, no. 5, pp. 803-838.

Keeney, R. L., 1984. Ethics, Decision Analysis, and Public Risk. Risk Analysis. Vol. 4, no. 2, pp. 117-129.

Keeney, R. L. & D von Winterfeldt, 1991. Eliciting Probabilities from Experts in Complex Technical Problems. IEEE Transactions on Engineering Management. Vol. 38, no. 3, pp. 191-201.

King, N. J., 1998. Application of Risk Assessment in Policy and Legislation in the European Union and the United Kingdom. In Handbook of Environmental Risk Assessment and Management. Chapter 9. Ed. Calow, P. Blackwell Science Ltd. Great Britain.

Kros, J., Reinds, G. J., De Vries, W., Latour, J. B. & M. Mollen, 1995. Modelling of soil acidity and nitrogen availability in natural ecosystems in response to changes in acid deposition and hydrology. Report 95. DLO Winand Staring Centre. Wageningen.

Kruber, S., Schoene, H., 1998. The application of decision analysis in the remediation sector. Environmental Engineering and Policy. No. 1, pp. 25-35.

Lepage N., Hamel, P., Lefebvre, R., Therrien, R. & C. Blais, 1999. Decision analysis for leachate control at a fractured rock landfill. Groundwater Monitoring and Remediation. Vol. 18, no. 3, pp 157-170

Levy, J. & E. E. Ludy, 2000. Uncertainty Quantification for Delineation of Wellhead Protection Areas Using the Gauss- Hermite Quadrature Approach. Ground Water. Vol. 38, no. 1, pp. 63-75.

Loaiciga, H. A. & M. A. Mariño, 1987. Parameter Estimation in Groundwater: Classical, Bayesian, and Deterministic Assumptions and Their Impact on Management Policies. Water Resources Research. Vol. 23, no. 6, pp. 1027-1035.

Maddock, III, T., 1973. Management Model as a Tool for Studying the Worth of Data. Water Resources Research. Vol. 9, no. 2, pp. 270-280.

Marin, C. M., 1986. Parameter Estimation in Water Resources Planning and Management: Optimal Actions or Optimal Parameters? Water Resources Research. Vol. 22, no. 3, pp. 353-360.

Marin, C. M., Medina, M. A., Jr. & J. B. Butcher, 1989. Monte Carlo analysis and Bayesian decision theory for assessing the effects of waste sites on groundwater, I: Theory. Journal of Contaminant Hydrology. No. 5, pp. 1-13.

de Marsily, G., 1986. Quantitative Hydrogeology. Groundwater Hydrology for Engineers. Academic Press, Inc. London.

Massmann, J. & R. A. Freeze, 1987a. Groundwater Contamination From Waste Management Sites: The Interaction Between Risk-Based Engineering Design and Regulatory Policy: 1. Methodology. Water resources research. Vol. 23, no 2, pp. 351-367.

Massmann,J. & R. A. Freeze, 1987b. Groundwater Contamination From Waste Management Sites: The Interaction Between Risk-Based Engineering Design and Regulatory Policy: 2. Results. Water resources research. Vol. 23, no 2, pp. 368-380.

Massmann, J., Freeze, R. A., Smith, L., Sperling, T. & B. James, 1991. Hydrogeological Decision Analysis: 2.Applications to Ground-Water Contamination. Ground Water. Vol. 29, no. 4, pp. 536-548.

Maxwell, R. M. & W. E. Kastenberg, 1999. Stochastic environmental risk analysis: an integrated methodology for predicting cancer risk from contaminated groundwater. Stochastic Environmental Research and Risk Assessment. Vol. 13, pp. 27-47.

Medina, M. A., Butcher, Jr., J. B. & C. M. Marin, 1989. Monte Carlo analysis and Bayesian decision theory for assessing the effects of waste sites on groundwater, II: Applications. Journal of Contaminant Hydrology. No. 5, pp. 15-31.

Mohamed, A. M. O. & K. Côté, 1999. Decision analysis of polluted sites – a fuzzy set approach. Waste Management. No. 19, pp. 519-533.

Mylopoulos, Y. A., Theodosiou, N. & N. A. Mylopoulos, 1999. A Stochastic Optimization Approach in the Design of an Aquifer Remediation under Hydrogeologic Uncertainty. Water Resources Management. Vol. 13, no. 5, pp. 335-351.

National Research Council (NRC), 1994. Science and Judgement in Risk Assessment. Committee on Risk Assessment of Hazardous Air Pollutants, Board on Environmental Studies and Toxicology, Commission on Life Sciences. National Academy of Sciences, Washington D. C.

National Research Council, 1996. Understanding Risk. Informing Decisions in a Democratic Society. National Academy Press. Washington D.C.

National Research Council, 1997. Valuing groundwater. Economic concepts and approaches. National Academy Press. Washington D.C.

Nikolaidis, N. P, Hellerich, L. A., & J. A Lackovic, 1999. Methodology for Site-Specific, Mobility-Based Cleanup Standards for Heavy Metals in Glaciated Soils. Environmental Science & Technology. Vol. 33, No. 17, pp 2910-2916.

Norman, F., 1999. Sweden. In Risk Assessment for Contaminated Sites in Europe: Volume 2. Policy Frameworks. Eds. Ferguson C. C. & H. Kasamas. LQM Press. Nottingham.

Olsson, L., 2000. Att bestämma subjektiva sannolikheter. (In Swedish). Varia 488. Swedish Geotechnical Institute. Linköping.

Olsson, O., Bäckblom, G., Gustafson, G., Rhén, I., Stanfors, R. & P. Wikberg, 1994. The structure of conceptual models with application to the Äspö HRL Project. Technical Report 94-08. SKB. Stockholm.

Paleologos, E. K. & I. Lerche, 1999. Multiple decision-making in the transport and burial of hazardous and radioactive wastes. Stochastic Environmental Research and Risk Assessment. Vol. 13, no. 6, pp. 381-395.

Palmer, R., 1998. Integration of Environmental and Economic Assessment. In Pollution Risk Assessment and Management, pp. 377-397. Ed. P. E. T. Douben. John Wiley & Sons Ltd, Chichester.

Pearce, D. W. & T. Seccombe-Hett, 2000. Economical Valuation and Environmental Desicion-Making in Europe. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1419-1425.

Press, J., Söderqvist, T., 1996. On estimating the benefits of groundwater protection: A contingent valuation study in Milan. NOTA DI LAVORO 53.96. Fondazione Eni Enrico Mattei. Milano.

Renn, O., 1999. A Model for an Analytic-Deliberative Process in Risk Management. Environmental Science & Technology. Vol. 33, no. 18, pp 3049-3055.

Richards, D. & W. D. Rowe, 1999. Decision-Making with Heterogeneous Sources of Information. Risk Analysis. Vol. 19, no. 1, pp. 69-81.

Rosén, L. & H. E. LeGrand, 1997. An Outline of a Guidance Framework for Assessing Hydrogeological Risks at Early Stages. Ground Water. Vol. 35, no. 2, pp. 195-204.

Rowe, W. D., 1994. Understanding Uncertainty. Risk Analysis. Vol. 14, no. 5, pp. 743-750.

Russell, K. T. & A. J. Rabideau, 2000. Decision Analysis for Pump-and-Treat Design. Groundwater Monitoring and Remediation.Vol. 19, no. 3, pp. 159-168.

Sagoff, M., 2000. Environmental Economics and the Conflation of Value and Benefit. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1426-1432.

Sandia National Laboratories, 1998. SmartSampling. A Tutorial: A real-life adventure in environmental decision making. From http://www.sandia.gov.

Sandström, M., 1998. Vad är vattnet värt? –Värderingsmodell för grundvattentillgångar. (In Swedish). Rapport 4876. Naturvårdsverket. Stockholm.

Selroos, J-O., 1997a. A Stochastic-Analytical Framework for Safety Assessment of Waste Repositories: 1. Theory. Ground Water. Vol. 35, no. 3, pp. 468-477.

Selroos, J-O., 1997b. A Stochastic-Analytical Framework for Safety Assessment of Waste Repositories: 2. Application. Ground Water. Vol. 35, no. 5, pp. 775-785.

Sharefkin, M., Schechter, M. & A. Kneese, 1984. Impacts, costs, and techniques for mitigation of contaminated groundwater: a review. Water resources research. Vol. 20, no 12, pp. 1771-1783.

Smrchek, J. C. & M. G. Zeeman, 1998. Assessing Risks to Ecological Systems from Chemicals. In Handbook of Environmental Risk Assessment and Management. Ed. P. Calow. Pp. 24-90. Blackwell Science. Great Britain.

Spash, C. L., 2000. Multiple Value Expression in Contingent Valuation: Economics and Ethics. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1433-1438.

Sperling, T., Freeze, R. A., Massmann, J., Smith, L. & B. James, 1992. Hydrogeological Decision Analysis: 3. Application to Design of a Ground-Water Control System at an Open Pit Mine. Ground Water. Vol. 30, no. 3, pp. 376-389.

Statens Naturvårdsverk, 1996a. Generella riktvärden för förorenad mark: Beräkningsprinciper och vägledning för tillämpning. Efterbehandling och sanering. (In Swedish). Rapport 5638. Stockholm.

Statens Naturvårdverk, 1996b. Development of generic guideline values. Model and data used for generic guideline values for contaminated soils in Sweden. Report 4639. Stockholm.

Statens Naturvårdsverk, 1997. Ekonomisk värdering av miljön. Metoder att mäta efterfrågan på miljötillgångar. (In Swedish). Rapport 4827. Stockholm.

Statens Naturvårdsverk, 1999a. Val av åtgärder: Metod för sammanvägning och samhällsekonomiska bedömningar. (In Swedish). Rapport 5005. Stockholm

Statens Naturvårdsverk, 1999b. Metodik för inventering av förorenade områden. Bedömningsgrunder för miljökvalitet. Vägledning för insamling av underlagsdata. (In Swedish). Rapport 4918. Uppsala.

Taylor, A. C., 1993. Using Objective and Subjective Information to Develop Distributions for Probabilistic Exposure Assessment. Journal of Exposure Analysis and Environmental Epidemiology. Vol. 3, no. 3, pp. 285-298.

Vägverket (VV), 1998. Förorening av vattentäkt vid vägtrafikolycka. Hantering av risker med petroleumutsläpp. (In Swedish). Publ 1998:064. Borlänge.

Wentsel, R. S., 1998. Application of Risk Assessment in Policy and Legislation in North America. In Handbook of Environmental Risk Assessment and Management. Chapter 10. Ed. Calow, P. Blackwell Science Ltd. Great Britain.

Whitehouse, P. & N. Cartwright, 1998. Standards for Environmental Protection. In Pollution Risk Assessment and Management. Ed. Douben, P. E. T. John Wiley & Sons Ltd, Chichester.

Wladis, D., Rosén, L., Kros, H., 1999. Risk-based Decision Analysis of Atmospheric Emission Alternatives to Reduce Ground Water Degradation on the European Scale. Ground Water. Vol. 37, no. 6, pp. 818-826.

Wood, W. W., 2000. It’s the Heterogeneity! Ground Water. Editorial note. Vol. 38, no. 1, pp 1.

APPENDIX A - RISK ASSESSMENT METHODS

Table A.1.[a] *Categorisation of principal risk assessment methods according to Covello and Merkhofer (1993).* [b]

RELEASE ASSESSMENT	EXPOSURE ASSESSMENT	CONSEQUENCE ASSESSMENT	RISK ESTIMATION
Monitoring ▪ Release monitoring ▪ Monitoring source status ▪ Monitoring administrative records ▪ Laboratory analysis **Performance testing** ▪ Component and system failure tests ▪ Accelerated-life tests ▪ Accident simulations ▪ Stress analysis ▪ Mental movies **Accident investigation** ▪ Field investigation ▪ Laboratory investigation ▪ Accident reconstruction **Statistical methods** ▪ Actuarial risk assessment ▪ Named probability distributions ▪ Bayes's theorem ▪ Statistical sampling ▪ Regression analysis ▪ Extreme value theory ▪ Hypothesis testing	**Monitoring** ▪ Personal exposure monitors (PEMs) ▪ Media contamination (site monitoring) - *Air, surface water, sediment, soil, groundwater* ▪ Remote geologic monitoring - *Aerial photography, multispectral overhead imegary* ▪ Biologic monitoring - *Chemical residues, bioaccumulation, biodegradation, physiology, indicator species* **Testing** ▪ Scale models ▪ Laboratory tests ▪ Field experimentation **Calculation of dose** ▪ Based on exposure time ▪ Coexisting or decay substances ▪ Material deposition in tissue	**Health surveillance** **Hazard screening** ▪ Molecular structure analysis ▪ Short-term tests **Animal tests** ▪ Acute toxicity studies ▪ Subchronic toxicity studies ▪ Chronic toxicity studies **Tests on humans** ▪ Laboratory setting ▪ Field setting **Epidemiology** ▪ Case-control study ▪ Cohort study ▪ Retrospective study ▪ Prospective study ▪ Molecular epidemiology **Animal-to-human extrapolation models** **Dose-response models** ▪ Threshold ▪ Tolerance ▪ Mechanistic ▪ Time-to-response **Pharmacokinetic models** **Ecosystem monitoring**	**Relative risk models** **Model coupling** **Risk indexes** ▪ Individual risk ▪ Societal risk **Nominal risk outcomes** **Worst-case outcomes** **Sensitivity analysis** ▪ Point ▪ Parametric ▪ Rank correlations ▪ Stochastic ▪ Closed loop **Statistical methods** **Probability encoding** ▪ Debiasing ▪ Interval method ▪ Probability wheel ▪ Behavorial aggregation ▪ Mechanical aggregation **Uncertainty propagation** ▪ Method of moments ▪ Monte Carlo analysis ▪ Response surfaces ▪ Probability trees **Quantitative uncertainty analysis** ▪ Confidence bounds ▪ Credibility analysis ▪ Uncertainty partitioning **Qualitative uncertainty analysis**

[a] The table is continued on the next page.
[b] For a complete explanation of all the methods the reader is referred to Covello and Merkhofer (1993).

Table A.1 continued.

RELEASE ASSESSMENT	EXPOSURE ASSESSMENT	CONSEQUENCE ASSESSMENT	RISK ESTIMATION
Modeling methods ▪ Engineering failure analysis ▪ Logic trees, event trees, fault trees, Markov models ▪ Analytic process models ▪ Biological models for pests ▪ Containment models ▪ Discharge models ▪ BLEVE models	**Pollutant transport-and-fate modelling** ▪ Air - *Analytic models, trajectory models, transformation models* ▪ Surface water - *Dissolved oxygen models, etc.* ▪ Groundwater - *Travel time models, absorption models* ▪ Overland ▪ Food-chain models ▪ Multimedia models **Exposure-route models** **Population-at-risk models** ▪ Census, sensitive groups, trip-generation models, etc.	**Tests on the natural environment** ▪ Field tests ▪ Laboratory tests ▪ Microcosms, macrocosms, mesocosms **Ecological effects models** ▪ Dynamic ▪ Matrix ▪ Stochastic ▪ Markov ▪ Harvest Pollution response	

References

Covello, V. T. & M. W. Merkhofer, 1993. Risk assessment methods: Approaches for assessing health and environmental risks. Plenum Press. New York.

APPENDIX B - INTERNATIONAL NETWORKS FOR CONTAMINATED LAND

Below are some international networks on contaminated land listed. Further information is available with Bardos et al. (1999) and Kasamas et al. (2000).

- CARACAS (Concerted Action on Risk Assessment for Contaminated Sites in Europe) was initiated to co-ordinate the efforts done with developing frameworks and procedures for assessing and managing risks. The main objective of CARACAS is to co-ordinate current research initiatives on contaminated land risk assessment carried out by the European Union Member States (http://www.caracas.at).
- NICOLE (Network for Industrially Contaminated Land in Europe) was created as a forum to bring problem holders and researchers together. It aims to identify research needs and promote co-ordinated, multidisciplinary, collaborative research that will enable European industry to identify, assess and manage contaminated sites more efficiently and cost-effectively. Additionally, NICOLE aims to inform relevant EU and Member States research planners of the needs and priorities for future research (http://www.nicole.org).
- CLARINETS (Contaminated Land Rehabilitation Network for Environmental Technologies) primary objective is to develop technical recommendations for the sound decision making for rehabilitation of contaminated sites in Europe. Academics, government experts, consultants, industrial landowners and technology developers represent sixteen European countries.
- The NATO/CCMS Pilot Study (Evaluation of Demonstrated and Emerging Technologies for the Treatment of Contaminated Land and Groundwater - Phase III) was designed to identify and evaluate innovative, emerging and alternative remediation technologies and to transfer technical performance and economic information on them to decision makers and potential users.
- NATO/CCMS Pilot Study: Environmental Aspects of Reusing Former Military Lands: One of its aim is to facilitate cooperation between NATO and former Eastern Bloc countries.
- Ad Hoc International Working Group for Contaminated Land: Initiated in 1993, with representatives from environmental ministries and agencies in 20 different countries together with international organisations such as FAO and OECD.
- Common Forum for Contaminated Land in the European Union - resulted in the Concerted Action CARACAS.
- RACE - Risk Abatement Center for contaminated soil in the CEE countries - was initiated in Poland with the aim to contribute to the development of risk-based standards for soil and groundwater in CEE countries, which will be acceptable for the EU.
- European Topic Centre on Soil (ETC/S). Its objective is to provide and develop information and data about soil conditions and status in all European countries.
- ISO Technical Committee (TC) 190/SC 7 Soil Quality - Soil and Site assessment. Established in 1995 with the aim to prepare international standards for reuse of soil material, for assessment of possible groundwater impact due to contaminated soil, ecotoxicological aspects of soil investigations and possible effects caused by human exposure.

Identified research needs

CARACAS and NICOLE produced a joint statement on identified research needs. Bardos, Kasamas and Denner (1999) claim the joint statement especially interesting as it represents a consensus approached from two different viewpoints: industry (NICOLE) and regulatory (CARACAS). The statement is available at http://www.caracas.at.

The Nature of Contaminated Land

A) Site characterisation: extent, intensity and environmental transport and fate of pollution

- robust and rapid low-cost techniques for investigation of potentially contaminated sites
- improved methods for estimating the accuracy and variability of the whole sampling and analytical process
- methods that yield information at spatial scales relevant for exposure assessment
- characterisation by biosensors and bioassays
- methods to assess migration of groundwater contamination
- methods to assess the natural potential of soil to reduce contaminants to acceptable risk levels and to monitor the process
- the interaction and general fate of contaminant mixtures
- detection of non-aqueous phase liquids and the prediction of their fate.

B) Bioavailability of contaminants in soil and groundwater

- to study the interaction between organisms (soil fauna, bacteria, plants) and their chemical environment
- time dependence (ageing) of bioavailability
- cost effective procedures for estimating bioavailable fractions in the environment.

Fitness for Use

A) Human health risks

- validation of human exposure pathways
- availability of contaminants within the human body
- availability of contaminants in the soil as compared to the availability in the animal experiments underlying most toxicological reference values.

B) Ecological risk assessment

- impact of a site on its environment
- ecological recovery at the site
- changes in community structure caused by pollution-induced tolerance versus classical ecotoxicological endpoints
- biomagnification and adverse effects on food chains
- ecological soil quality requirements related to human land use.

C) Risk perception and communication

- Risk perception of contaminated land
- Development of communication strategies: how to communicate the results of risk assessments and the choice of solutions to those potentially at risk and to other interested parties.

D) Remediation Technologies

- Processes of natural attenuation
- Low-energy approaches
- Cost-effective remedial technologies
- Monitoring of remediation

References

Bardos, P., Kasamas, H., & J. Denner, 1999. International networks for contaminated land. In Risk Assessment for Contaminated Sites in Europe. Volume 2. Policy Frameworks. Eds. Ferguson C. C. & H. Kasamas. LQM Press. Nottingham.

Kasamas, H., Denner, J., Skovgard, I-M. & P. Bardos, 2000. International Networks on Contaminated Land – An Overview. http://www.caracas.at. 2000-03-03.

APPENDIX C. SUGGESTED FURTHER READING

Abbaspour, K. C., Schulin, R., Schläppi, E. & H. Flühler, 1996. A Bayesian approach for incorporating uncertainty and data worth in environmental projects. Environmental Modelling and Assessment. Vol. 1, no 3/4, pp. 151-158.

Ahlfeld, D. P., Mulvey, J. M. & G. F. Pinder, 1988. Contaminated Groundwater Remediation Design Using Simulation, Optimisation, and Sensitivity Theory. 1. Model Development. Water Resources Research. Vol. 24, no. 3, pp. 431-441.

Arrow, K., Daily, G., Dasgupta, P., Levin, S., Mäler, K-G., Maskin, E., Starrett, D., Sterner, T. & T. Tietenberg, 2000. Managing Ecosystem Resources. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1401-1406.

Ben-Zvi, M., Berkowitz, B. & S. Kessler, 1988. Pre-posterior Analysis as a Tool for Data Evaluation: Application to Aquifer Contamination. Water resources management. Vol. 2, pp. 11-20.

Bockstael, N. E., Freeman, A. M. III, Kopp, R. J., Portney, P. R. & V. K. Smith, 2000. On Measuring Economic Values for Nature. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1384-1389.

Bogárdi, I., Bardossy, A. & L. Duckstein, 1985. Multicriterion Network Design Using Geostatistics. Water Resources Research. Vol. 21, no. 2, pp. 199-208.

Bogen, K., 1995. Methods to Approximate Joint Uncertainty and Variability in Risk. Risk analysis. Vol. 15, no. 3, pp. 411-419.

Carson, R. T., 2000. Contingent Valuation: A User's Guide. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1413-1418.

Cothern, C. R. (ed.), 1996. Handbook for Environmental Risk Decision Making. Values, Perceptions, & Ethics. CRC Press, Inc. Florida.

Dagan, G, 1986. Statistical Theory of Groundwater Flow and Transport. Pore to Laboratory, Laboratory to Formation, and Formation to Regional Scale. Water Resources Research. Vol. 22, no. 9, pp. 120S-134S.

Dagan, G., 1982. Stochastic Modeling of Groundwater Flow by Unconditional and Conditional Probabilities. 1. Conditional Simulation and the Direct Problem. Water Resources Research. Vol. 18, no. 4, pp. 813-833.

Dagan, G., 1985. Stochastic Modeling of Groundwater Flow by Unconditional and Conditional Probabilities. 3. The Inverse Problem. Water Resources Research. Vol. 21, no. 1, pp. 65-72.

Delhomme, J. P., 1979. Spatial Variability and Uncertainty in Groundwater Flow Parameters. Water Resources Research. Vol. 15, no. 2, pp. 269-280.

Einstein, H. H. & G. B. Baecher, 1983. Probabilistic and Statistical Methods in Engineering Geology. Specific Methods and Examples. Part I: Exploration. Rock Mechanics and Rock Engineering. Vol. 16, pp. 39-72.

Farber, S. & B. Griner, 2000. Using Conjoint Analysis to Value Ecosystem Change. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1407-1412.

Fischoff, B., 2000. Informed Consent for Eliciting Environmental Values. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1439-1444.

Freeze, R. A., 1975. A Stochastic-Conceptual Analysis of One Dimensional Groundwater Flow in Nonuniform Homogeneous Media. Water Resources Research. Vol. 11, no. 5, pp. 725-741.

Gelhar, L. W.,1986. Stochastic Subsurface Hydrology From Theory To Applications. Water Resources Research. Vol. 22, no. 9, pp. 135S-145S.

Goicoechea, A., Duckstein, L. & M. M. Fogel, 1979. Multiple Objectives Under Uncertainty: An Illustrative Application of Protrade. Water Resources Research. Vol. 15, no. 2, pp. 203-210.

Goodrich, M. T. & J. T. McCord, 1995. Quantification of Uncertainty in Exposure Assessment at Hazardous Waste Sites. Ground Water. Vol. 33, no. 5, pp. 727-732.

Gorelick, S. M., 1982. A Model For Managing Sources of Groundwater Pollution. Water Resources Research. Vol. 18, no. 4, pp. 773-781.

Hachich, W. & E. H. Vanmarcke, 1983. Probabilistic Updating of Pore Pressure Fields. Journal of Geotechnical Engineering. Vol. 109, no. 3, pp. 373-385.

Haimes, Y. Y., 1983. Integrated risk and uncertainty assessment in water resources within a multiobjective framework. Journal of Hydrology. Vol. 68, pp. 405-417.

Hansson, S-O., 1993. Decision-making when experts disagree. Translation of: Entscheidungsfindung bei Uneinigkeit der Experten. In Die Midernisierung der Demokratie, Internationale, Ansätze. Dienel, P. C. & W. Strubelt (eds.). Westdeutscher Verlag.

Hansson, S-O., 1993. The false promises of risk analysis. Ratio (New Series), Vol. VI, pp. 16-26.

Hansson, S-O., 1996. Decision Making Under Great Uncertainty. Philosophy of the Social Sciences. Vol. 26, no. 3, pp. 369-386.

Hansson, S-O., 1996. What is philosophy of risk? Theoria. Vol. 62, pp. 169-186.

Irish, J. L., 1993. The distribution of the present worth of risky outcomes. In Probabilistic Risk and Hazard Assessment. Melchers & Stewart (eds.). Balkema. Rotterdam.

Joos, W., Carabias, V., Winistoerfer, H. & A. Stuecheli, 1999. Social aspects of public waste management in Switzerland. Waste Management. Vol. 19, pp. 417-425.

Journel, A. G., 1983. Nonparametric Estimation of Spatial Distributions. Mathematical Geology. Vol. 15, no. 3, pp. 445-468.

Journel, A. G., 1986. Constrained Interpolation and Qualitative Information – The Soft Kriging Approach. Mathematical Geology. Vol. 18, no. 3, pp. 269-286.

Journel, A. G., 1986. Geostatistics: Models and Tools for the Earth Sciences. Mathematical Geology. Vol. 18, no. 1, pp. 119-140.

Kool, J. B., Parker, J. C. & M. TH. van Genuchten, 1987. Parameter estimation for unsaturated flow and transport models-a review. Journal of Hydrology. Vol. 91, pp. 255-293.

Krzysztofowicz, R., 1986. Expected Utility, Benefit, and Loss Criteria for Seasonal Water Supply Planning. Water Resources Research. Vol. 22, no. 3, pp. 303-312.

Lee, D. A. & J. M. Browne, 1993. Implications of varying probability in fault tree analysis. In Probabilistic Risk and Hazard Assessment. Melchers & Stewart (eds.). Balkema. Rotterdam.

LeGrand, H. E. & L. Rosén, 1992. Common Sense in Ground-Water protection and management in the United States. Ground Water. Vol. 30, no. 6, pp. 867-872.

LeGrand, H., 1982. Hydrogeological Issues in Risk Assessment of Hazardous Waste Sites. Ground Water. Vol. 20, pp. 488-491.

Loaiciga, H. A., Charbeneau, R. J., Everett, L. G., Fogg, G. E., Hobbs, B. F. & S. Rouhani, 1992. Review of Ground-Water Quality Monitoring Network Design. Journal of Hydraulic Engineering. Vol. 118, no. 1, pp. 11-37.

Mahar, P. S. & B. Datta, 1997. Optimal Monitoring Network and Ground-Water-Pollution Source Identification. Journal Of Water Resources Planning and Management. Vol. 123, no. 4, pp. 199-207.

Marsily, G. de, Delay, F., Teles, V. & M. T. Schafmeister, 1998. Some current methods to represent the heterogeneity of natural media in hydrogeology. Hydrogeology Journal. Vol. 6, pp. 115-130.

Matthew, H. S. & L. B. Lave, 2000. Applications of Environmenatl Valuation for Determining Externality Costs. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1389-1395.

Maxwell, R. M., Pelmulder, S. D., Tompson, A. F. B. & W. E. Kastenberg, 1998. On the development of a new methodology for groundwater driven health risk assessment. Water Resources Research. Vol. 34, no. 4, pp. 833-847.

McDonald, J. F., 1981. The Use of Weighted Discount Rates in Cost-Benefit Analysis: A Further Analysis. Water Resources Research. Vol. 17, no. 3, pp. 478-480.

McKone, T. E. & K. T. Bogen, 1991. Predicting the uncertainties in risk assessment. A California groundwater case study. Environmental Science and Technology. Vol. 25, no. 10, pp. 1674-1681.

Melchers, R. E., 1993. On the treatment of uncertainty information in PRA. In Probabilistic Risk and Hazard Assessment. Melchers & Stewart (eds.). Balkema. Rotterdam.

Melchers, R. E., 1993. Society, tolerable risk and the ALARP principle. In Probabilistic Risk and Hazard Assessment. Melchers & Stewart (eds.). Balkema. Rotterdam.

Mourato, S., Ozdemiroglu, E. & V. Foster, 2000. Evaluating Health and Environmental Impacts of Pesticide Use: Implications for the Design of Ecolabels and Pesticide Taxes. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 1456-1461.

Poeter, E. P. & S. A. Mckenna, 1995. Reducing Uncertainty Associated with Ground-Water Flow and Transport Predictions. Ground Water. Vol. 33, no. 6, pp. 899-904.

Poulos, C. & D. Whittington, 2000. Time Preferences for Life-Saving Programs: Evidence from Six Less Developed Countries. Environmental Science and Technology. Special issue: Valuing the Environment. Vol. 34, no. 8, pp. 14445-1455.

Raucher, R. L., 1984. A Conceptual Framework For Measuring The Benefits of Groundwater Protection. Water Resources Research. Vol. 19, no. 2, pp. 320-326.

Reichard, E. G. & J. S. Evans, 1989. Assessing the Value of Hydrogeological Information for Risk-Based Remedial Action Decisions. Water Resources Research. Vol. 25, no. 7, pp. 1451-1460.

Rouhani, S. & T. J. Hall, 1988. Geostatistical schemes for groundwater sampling. Journal of Hydrology. Vol. 103, no. 1/2, pp. 85-102.

Rouhani, S., 1985. Variance Reduction Analysis. Water Resources Research. Vol. 21, no. 6, pp. 837-846.

Sánchez-Vila, X. & J. Solís-Delfin, 1999. Solute transport in heterogeneous media: The impact of anisotropy and non-ergodicity in risk assessment. Stochastic Environmental Research and Risk Assessment. Vol. 13, pp. 365-379.

Smith, L. & F. W. Schwartz, 1981**.** Mass Transport. 2. A Stochastic Analysis of Uncertainty in Prediction. Water Resources Research. Vol. 17, no. 2, pp. 351-369.

Starr, C., Rudman, R. & C. Whipple, 1976. Philosophical basis for risk analysis. Annual Review of Energy. Vol. 1, pp. 629-662.

Sturk, R., 1998. Engineering Geological Information – It's Value and Impact on Tunnelling. Doctoral Thesis 1027. Division of soil and Rock Mechanics. Department of Civil and Environmental Engineering. Royal Institute of Technology. Stockholm. Sweden.

Tucciarelli, T. & G. Pinder, 1991. Optimal Data Acquisition Strategy for the Development of a Transport Model for Groundwater Remediation. Water Resources Research. Vol. 27, no. 4, pp. 577-588.

Wagner, J. M., Shamir, U. & H. R. Nemati, 1992. Groundwater Quality Management Under Uncertainty: Stochastic Programming Approaches and the Value of Information. Water Resources Research. Vol. 28, no. 5, pp. 1233-1246.

Wang, M. & C. Zheng, 1997. Optimal Remediation Policy Selection under General Conditions. Ground Water. Vol. 35, no. 5, pp. 757-764.

Whitman, R. V., 1984. Evaluating Calculated Risk in Geotechnical Engineering. Journal of Geotechnical Engineering. Vol. 110, no. 2, pp. 145-188.

Whitman, R. V., 2000. Organizing and Evaluating Uncertainty in Geotechnical Engineering. Journal of Geotechnical and Geoenvironmental Engineering. Vol. 126, no. 7, pp. 583-593.

Wolka, K. K. & T. A. Austin, 1988. Groundwater Contamination Benefit-Cost Analysis Methodology. Journal of Water Resources Planning and Management. Vol. 114, no. 2, pp. 210-222.