THE DEVELOPMENT OF A WEB-BASED DECISION SUPPORT SYSTEM FOR THE SUSTAINABLE MANAGEMENT OF CONTAMINATED LAND

Submitted by

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I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

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ABSTRACT

Land is a finite natural resource that is increasingly getting exhausted as a result of land contamination. Land is made up of soil and groundwater, both of which have many functions for which we depend on, including provision of food and water, supporting shelter, natural flood defence, carbon sequestration, *etc*. Contaminants in land also pose a number of threats to public health and the environment; other natural resources; and have detrimental effects on property such as buildings, crops and livestock. The most effective method of dealing with these contaminants is to cleanup and return the sites to beneficial use. The cleanup process involves making a choice from amongst competing remediation technologies, where the wrong choice may have disastrous economic, environmental and/or social impacts. Contaminated land management is therefore much broader than the selection and implementation of remedial solutions, and requires extensive data collection and analysis at huge costs and effort.

The need for decision support in contaminated land management decision-making has long been widely recognised, and in recent years a large number of Decision Support Systems (DSS) have been developed. This thesis presents the development of a Web-based knowledge-based DSS as an integrated management framework for the risk assessment of human health from, and sustainable management of, contaminated land. The developed DSS is based on the current UK contaminated land regime, published guidelines and technical reports from the UK Environment Agency (EA) and Department for Environment, Food and Rural Affairs (DEFRA) and other Government agencies and

departments. The decision-making process of the developed DSS comprises of key stages in the risk assessment and management of contaminated land: (*i*) preliminary qualitative risk assessment; (*ii*) generic quantitative risk assessment; *and* (*iii*) options appraisal of remediation technologies and remediation design. The developed DSS requires site specific details and measured contaminant concentrations from site samples as input and produces a site specific report as output. The DSS output is intended to be used as information to support with contaminated land management decision-making.

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1 INTRODUCTION

1.1 PREAMBLE

Natural resources such as land are finite and are exponentially getting exhausted as a result of land contamination. Contaminants in land pose a number of threats to public health and the environment, including the health and safety of those working on or living near contaminated sites; other essential natural resources such as air and water; and detrimental effects on property such as buildings, crops and livestock. The most effective method of dealing with the contaminants is to remediate and return the site to beneficial use by either: (*i*) removing the contaminant sources; (*ii*) treating the contaminants to reduce or eliminate harm; *or* by (*iii*) containing the contaminants to isolate them. Contaminated land management is therefore much broader than the selection and implementation of remedial solutions, requiring extensive data collection and analysis at huge costs and effort (Vegter 2001). Management decision-making therefore involves making a choice from amongst competing alternative courses of action where the wrong outcome may have disastrous economic, environmental or social impacts (Sànchez-Marrè et al 2008).

Although current policy is well developed and there is good scientific and technical understanding of the nature and extent of land contamination and the behaviour of contaminants in the sub-surface environment, effective management is challenging as it relies on good understanding and application of a vast multidisciplinary knowledge bases that straddle the natural, physical, engineering and social sciences within technologically practical, economically viable, and regulatory constraints (Pollard et al 2001). These challenges include (*after* Hester and Harrison 1997, Rulkins et al 1998, Pollard et al 2001, Bardos et al 2001):

- Differences in the nature, extent, concentrations and heterogeneity of contaminants present at each site, and the difficulty in establishing contamination boundaries.
- Individual site characteristics such as hydrogeology, hydrology, geology, land use type, and other specific aspects of the each site itself.
- Site specific uncertainties resulting from the complexity of the soil environment due to multiple interactions and feedbacks of land systems.
- The need for integrating vast multidisciplinary knowledge bases involving different people from different areas of expertise such as engineers, geologists, environmental and chemical analysts, each involved in interpreting discipline specific information for decision-making in a useful format.
- There is often incomplete knowledge as full information is rarely available or attainable, resulting in further uncertainties in risk assessment and ultimately confidence in the decision-making process.
- There is also a range of contexts in which decisions have to be made, such as compliance with the relevant legislative framework, assessment of total operating costs and benefits, environmental impacts on other resources, and addressing issues of sustainability and environmental stewardship.

Additionally contaminated land practitioners are increasingly facing threats of financial liability in cases of ineffective solutions. These challenges are not new and have been recognised by policy makers internationally for at least three decades (Pollard et al 2001).

As a result of this attempts have been made to structure the vast multidisciplinary and specialist expertise into coherent decision-making frameworks, in an effort to reduce the complexity and uncertainty, and reduce management costs and time scales. Despite these developments however, a majority of these decision-making frameworks have not been translated into software applications (CLARINET 2002*b*).

1.2 GAPS AND CHALLENGES

Numerical and statistical models that are very advanced in both development and practical application have long been used to provide information to support contaminated land decision-making. However models alone cannot fully deliver all the functionalities envisaged for decision support (Rizzoli and Young 1997). This is because not all environmental systems present the same level of complexity in terms of both the degree of uncertainty and the risk associated with decisions. Three levels of complexity have been established with environmental management decision-making (Sànchez-Marrè et al 2008):

i. The first level of complexity involves simple, low uncertainty, limited scope problems, where models can sufficiently provide satisfactory problem descriptions.
ii. The second level of complexity involves problems with a higher degree of uncertainty where models alone cannot provide satisfactory problem descriptions, and expert knowledge is often required to support decision-making.
iii. The third level of complexity involves truly complex systems, where much epistemological or ethical uncertainty exists, with issues at stake reflecting conflicting goals, and support systems are often needed.

Contaminated land management decision-making problems characteristically involve the second and third levels of complexity, requiring the use of new methods and tools to adequately support decision-making. The need for decision support in contaminated land management decision-making is widely recognised (CLARINET 2002*b*), and increasingly Geographical Information Systems (GIS), Decision Support Tools (DST) and Decision Support Systems (DSS) have been used to support contaminated land management decision-making. Standalone GIS have been used to support different aspects of contaminated land management, *e.g*. development of spatial site conceptual models for spatial analysis of pollutant linkages or risk assessment. However although standalone GIS can provide some decision support capabilities they lack the capability for complex analysis of unstructured problems (Segrera et al 2003), and also do not provide a means for assessing and choosing from amongst competing alternatives. As such standalone GIS cannot be used to adequately support decision-making (Yan et al 1999).

DSS are amongst the most promising solutions because of their ability to integrate different frameworks, architectures, tools and methods for solving high level complexity (Poch et al 2003). Although one may argue that a model or GIS could be used for decision support, today's consensus is that environmental DSSs (EDSSs) must adopt a knowledge-based approach (*after* Poch et al 2003), which can easily be implemented in DSS. In recent years a large number of DSTs and DSSs have been developed for integrating the vast multidisciplinary knowledge-bases into coherent frameworks to support consistent, rational and transparent decision-making that is reproducible and therefore justifiable (Bardos et al 2001). Many DSS have been developed for supporting contaminated land management decision problems with varying degrees of success in practical application (CLARINET

2002*a*), *e.g*. Jankwowski and Stakis 1997, Chen and Chakma 2002, Mustajoki et al 2004, Birkin et al 2006, *etc*. These have been presented in different formats from simple diagrams derived from standards or regulations, to computer-based models that simulate contaminant behavior and transport, to DSTs and DSSs supporting high level management decision-making. These have varied from straightforward information about pros and cons of remedial options to formalised weighting systems, with varying degrees of success in practical application (Vegter 2001, CLARINET 2002*b*).

A lot of these DSS are models or tools used for data visualisation or system description however, and do not specifically address decision problems or help decision makers in making inevitable trade-offs (Giove et al 2008). The majority of these DSS also focus on risk assessment, technology selection or stakeholder involvement, and rarely address the overall contaminated land management process (Agostini and Vaga 2008). It is widely accepted that taking management decisions in isolation is no longer sufficient and there is a need for a robust and integrated decision-making framework (Pollard et al 2004), as all aspects of the management process are interrelated and have a bearing on the final decision outcome. There is therefore a need to integrate the different models and tools into single systems for effective management. Integration has been a challenge however, as different tools and models are developed using different methods and may be developed with different programming languages and/or on different architectures.

1.3 RESEARCH OBJECTIVES

The main objective of this research was the development of a DSS for the sustainable management of contaminated land. The DSS was intended to integrate different methods, tools and techniques to support timely and cost-effective management decision-making by ensuring that the management process is scientifically and technically sound, consistent and transparent. A transparent decision-making process is reproducible and therefore justifiable. To that effect a DSS has been developed as an integrated framework for the whole management life cycle, involving: (*i*) preliminary qualitative risk assessment; (*ii*) generic quantitative risk assessment; *and* (*iii*) options appraisal of remediation technologies and remediation design. The DSS provides a site report at the end of the management decision-making process as output, which can be used as information for decision support.

Due to the vast areas of research, information and data availability of contaminated land management and time limitation of the project, it was necessary to scope the DSS and a decision was made to develop a DSS for a specific type of contamination, as opposed to a generic management system. Problem specific DSS are tailored to specific environmental problems, but can be applied to a wide range of different locations with the same problem, and have a wide range of other advantages (Rizzoli and Young 1997). A component based approach to software development was used in developing the DSS because it was recognised that both the scientific understanding of contaminant behaviour and transport and the corresponding technical understanding of remediation are constantly evolving, resulting in changes with regulatory and clean-up requirements and ultimately the decision-making process. Additionally different aspects of the management decision-making process change at different times, requiring that the DSS be developed in such a way that different parts can be adapted without disrupting other system components.

The DSS was developed on an open source LAMP (Linux operating system, Apache HTTP server, MySQL database server and the PHP scripting language) server. The developed DSS consists of three core components: (*i*) a database component; (*ii*) a model component; *and* (*iii*) a User Interface (UI) component. The DSS was developed as a Web-based application, on an *n*-tier client-server architecture with the first tier as the presentation layer (the DSS UI), the second tier the application layer (the decision model), and the third tier the storage layer (the DSS database). The database was developed as a Relational Database (RDB) model, using the international standard database language, the Structured Query Language (SQL), embedded in MySQL database server. The decision model was developed as a Multi Criteria Decision Analysis (MCDA) model using the Analytical Hierarchy Process (AHP). The result of the decision model was encapsulated in a knowledge-base. The knowledge-base was developed using the using the CLIPS expert system shell. The DSS UI has been developed as a common interface between the different DSS components. The UI was developed as a Graphical User Interface (GUI) using mixed language programming paradigm, using: eXtensible HyperText Markup Language (XHTML), PHP: Hypertext Preprocessor, JavaScript, Asynchronous JavaScript and XML (AJAX) and Cascading Style Sheets (CSS).

1.4 THESIS OUTLINE

This thesis presents the development of a Web-based knowledge-based DSS as an integrated framework for the risk assessment of human health and management of petroleum hydrocarbon contamination, using the current UK contaminated land policy, the UK Environment Agency (EA) framework for risk assessment from petroleum hydrocarbon contamination in soils, and supporting statutory and non-statutory guidelines and technical reports prepared by the EA, the Department for Environment, Food and Rural Affairs (DEFRA) and other Government departments and agencies. The decision-making process of the DSS comprises of three stages:

i. Preliminary (qualitative) risk assessment which uses information collected during desk study and site investigation phase as input parameters for site specific characterisation. The characterisation is based on site end-use, neighbouring land uses, presence of water resources and the soil vulnerability of the site. The result of the characterisation is used for other stages of the decision-making process.
ii. Generic quantitative risk assessment (GQRA) which involves comparing the measured concentrations of site samples with Generic Assessment Criteria (GAC). The GAC values that are used for the GQRA in the DSS are based on published EA Soil Guideline Values (SGV) which are based on generic assumptions on contaminant fate and transport in the environment, generic conceptual site conditions and human behaviour to estimate child and adult exposures to soil contaminants for three generic land use scenarios (EA 2009). Where EA values are not available, the Land Quality Management/Chartered Institute of Environment Health (LQM/CIEH) GAC values have been used (LQM/CIEH 2009). The DSS also provides the option of comparing with Dutch Intervention Values as most site samples are analysed and assessed using DIV (DIV 2000).
iii. Options appraisal which is used for comparing remediation technologies based on selected sustainability criteria and sub-criteria to ensure that remediation is sufficient and proportional to requirements. The criteria available are based on sustainability indicators defined by the Sustainable Remediation Forum UK (SuRF-

UK) covering a range of economic, environmental and social issues (Bardos et al 2009). The remediation technologies used in the DSS and their technical criteria are based on information from the USA Environmental Protection Agency and other US departments (EPA *c*2010). The rationale for the decision-making process is based on published guidance and technical reports by DEFRA and the EA – CLR7: Assessment of Risks to Human Health from Land Contamination: An Overview of the Development of Soil Guideline Values and Related Research (DEFRA and EA 2002*a*); CLR8: Priority Contaminants Report (DEFRA and EA 2002*b*); CLR9: Contaminants in Soils: Collation of Toxicological Data and Intake Values for Human Health (DEFRA and EA 2002*c*); CLR10: Contaminated Land Exposure Assessment Model (CLEA): Technical Basis and Algorithms (DEFRA and EA 2002*d*); *and* CLR 11: Model procedures for the management of land contamination (DEFRA and EA 2002*e*).

The DSS generates a site specific report which covers site characterisation, risk assessment and options appraisal that can be used as information to support management decision-making. The report format uses EA guidelines for contaminated land reports – GPLC1: Guiding principles for land contamination (EA 2010*a*) and GPLC3: Reporting checklists (EA 2010*a*). The DSS reports are intended to provide a framework for rapid generation of scientific and technically sound information that is consistent, transparent and reproducible to support management decision-making. This is intended to reduce management costs and time and to offer increased confidence in management decision-making.

Due to the diversity and multidisciplinary nature of the research, the thesis is written mostly to cover the new ideas and the fundamental concepts needed to establish the common-grounds between the disciplines. This chapter covered a general introduction to land contamination, the management challenges contaminated land practitioners face, and the need for management decision support. The remainder of the thesis is organised as follows:

Chapter 2 overviews land contamination, its definition and scope within the UK legislative context, its extent and implications, policy drivers and the contaminated land management process. The chapter discusses the multi-dimensionality and complexity of the management decision-making process and the approaches and methods that are used to deal with it.

Chapter 3 reviews the different methods, tools and techniques that are used for supporting environmental decision-making and with specific references to the application of these technologies in supporting contaminated land management decision-making. These solutions are used on a case by case basis however, and as a result decision-making frameworks and DSSs are used to encapsulate these for automating the decision-making process for supporting similar management problems.

Chapter 4 overviews DSS technology, its characteristics, capabilities, taxonomy, architectural composition, and reviews its use in supporting contaminated land management decision-making. The chapter also reviews the different types of DSTs and DSSs that have been developed for contaminated land management.

Chapter 5 presents the development of a generic framework for developing contaminated land management DSS, which considers the DSS development life cycle; the characteristics, requirements and constraints of contaminated land management decision-making; the contaminated land management decision-making process; and the appropriate decision support tools and methods that can be used to support contaminated land management decision-making.

Chapter 6 presents the development of a DSS for contaminated land management based on the framework proposed in chapter 5. The DSS developed is a Web-based knowledge-based system for the risk assessment of human health and the sustainable management of petroleum hydrocarbon contamination, based on the current UK contaminated land legislation and regulatory requirements.

Chapter 7 presents the evaluation of the DSS in order to establish what the DSS knows, knows correctly, and/or what it does not know. This involved: (*i*) verification by testing and debugging the DSS source code of each component at each stage of the development process; *and* (*ii*) validating the appropriateness of the DSS in supporting real world contaminated land management decision problems using real life case studies in order to evaluate all aspects of design, development and practical application of the DSS.

Chapter 8 concludes the thesis by summarising the literature reviewed, the gaps and challenges identified, the work done and highlights the contributions of the research in supporting contaminated land management decision-making. The chapter concludes by providing recommendations for further work in the development and practical application of contaminated land management DSS.

Footnotes have been used throughout the thesis to explain definition of key terms and acronyms so as not to interrupt the flow of reading.

Appendices are provided at the end of the thesis:

Appendix I provides the GAC used in the DSS and their sources

Appendix II provides a description of the remediation technologies used in the DSS

Appendix III provides the sustainability criteria used in the DSS

Appendix IV covers the design, development and evaluation of the database component

Appendix V provides the design, development and validation of the decision model

2 CONTAMINATED LAND MANAGEMENT

2.1 INTRODUCTION

Land contamination is a major environmental and infrastructural problem in industrialised countries as a result of both past and present industrial processes and waste disposal activities. Land contamination could also occur naturally as part of the local geology or natural degregation. In the UK increasingly more Greenfields[1] that should be preserved and protected are being threatened and lost to development as a result of land contamination or dereliction, although there are numerous abandoned and derelict sites that could be sustainably regenerated and redeveloped for this purpose. Over 1,100 *ha* of UK greenfields have been lost to development each year since 1997 alone (CPRE 2008).

Contaminated lands not only cause loss of valuable land for food and housing but pose significant potential risks to human health and other receptors like water resources, ecosystems and infrastructure. Contaminants in land could affect human health through various exposure pathways such as inhalation of air, ingestion of food or dermal contact. These could be present in solid, liquid or gas phases, and may be physical, chemical or biological (Young et al 1997). Although rare, increased levels of illnesses has been observed on people living on or near lands affected by contamination – such as organ damage (BBC 2001), birth deformities (BBC 2009, Beck 1979) and cancers (Hansen et al 1997). Contaminants in soil can also pollute valuable water resources such as surface

[1] Greenfields are previously undeveloped land.

waters and groundwater aquifers (Powell et al 2003, Ford and Tellam 1994), ecological systems and habitats (Smith et al 2005), and pose other hazards such as fires and explosions on property (Brown and Maunder 1994, Harber and Forth 2002, Young et al 1997).

Land is made up of soil and groundwater which are both finite resources because they form and regenerate on geological timescales. Soil covers most of the earth's land surface, varying in depth from a few centimetres to several meters. Healthy soils interacts with air to maintain the balance of essential gases and regulate the drainage and flow of groundwater, thereby acting as a filter of contaminants and a natural flood defence. There are therefore vital links between the soil, air and groundwater; with soil acting as a buffer system and the link between these resources. Soil is also a large natural store of carbon, with UK soils alone containing around 10 billion tones. The loss of this is estimated to create emissions equivalent to more than 50 times the UK's current annual greenhouse gas emissions. Soil will therefore play a vital role in the fight against climate change (DEFRA 2009).

For the soil to perform these functions it must be healthy and managed effectively and sustainably. Unfortunately soil is a non-renewable resource as it can take up to hundreds or thousands of years to form through the different geologic processes and as such needs to be protected and preserved. Although evidence suggests that most sources of soil contamination are now suitably controlled, continued diffuse (*non-point source*) pollution[2] from atmospheric depositions, leaching and run-off is an area of growing concern (EA 2009). Diffuse pollution remains the main source of pollution of controlled water resources.

[2] Diffuse pollution occurs when the sources of the contamination are not known, and could arise from many different sources.

Controlled waters comprise of all estuaries; surface waters like streams, rivers and lakes; groundwater resources; and territorial waters[3]. Groundwater is the largest source of fresh water supply for which many people around the world depend. About 30 percent of this groundwater supply is bound up in ice and snow, with only about 0.2 percent available as freshwater in lakes and river. This freshwater supply is the primary source of drinking water for billions of people around the world. In the UK for example, it is a third of the total drinking water supply in England and Wales, and in parts of the South East of England the only source of drinking water (EA 2007).

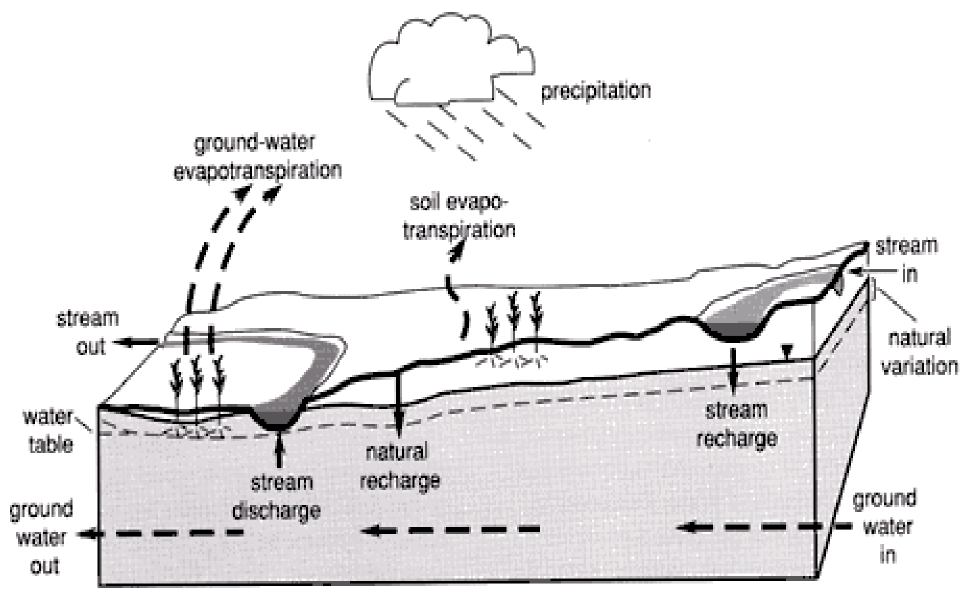

Figure 2.1 – The water cycle (Buchanan and Buddemeier 2005)

Groundwater is formed by the water cycle (*Fig 2.1*) when rainwater infiltrates into the sub-surface and is stored in soil pores and permeable geologic formations known as aquifers. This eventually flows to the surface naturally as surface water, thereby maintaining fresh

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[3] Territorial waters are coastal waters up to three nautical miles from shore.

water habitats. As groundwater can be a long-term water reservoir where the natural water cycle takes anything from days to millennia to complete, it is exposed to potential contamination by pollutants leaching and/or running-off from degraded or contaminated soil. Over two thirds of the groundwater in the UK is at risk from diffuse pollution, with pollutants from fertilisers, manure, pesticides, oil and fuel comprising the main sources of groundwater contamination (EA 2007). Grounwater is particularly vulnerable to diffuse pollution, which can take decades or centuries to recover because most chemicals degrade very slowly and groundwater is flushed through at a very slow rate. It is therefore a lot easier technically and more cost-effective to deal with point-source contamination, *i.e*. dealing with contamination in soil before the contaminants pollute water resources.

2.2 EXTENT OF LAND CONTAMINATION

Various national estimates have been made of how much contaminated land there is, which has varied considerably over the years as definitions and contexts evolved (Martin 2002). The UK Environment Agency (EA) estimate there may be as many as 200,000 *ha* affected by contamination in England and Wales alone (EA 2001) representing between 0.4 and 0.8 percent of the total UK land area (Young et al 1997). Between five and 20 percent of these are thought to require action to ensure that unacceptable risks to human health and the environment are minimised or eliminated (EA 2002). Conservative estimates say it will take between £20 – 40 billion to clean up and return these lands to beneficial use (Watson 1993). The figure is significantly increasing with the identification of more contamination.

However estimates of the extent of land contamination are often based on different definitions and terms which are not only fundamentally but technically different and therefore need to be viewed with caution (Pollard et al 2001). For example, the term Brownfield[4] is often used interchangeably with contaminated land, although current UK legislation does not refer to it. Even terms such as derelict land, Previously Developed Land (PDL), and land affected by contamination, that have been clearly defined are also often used interchangeably although they are all technically quiet different (*Fig 2.2*).

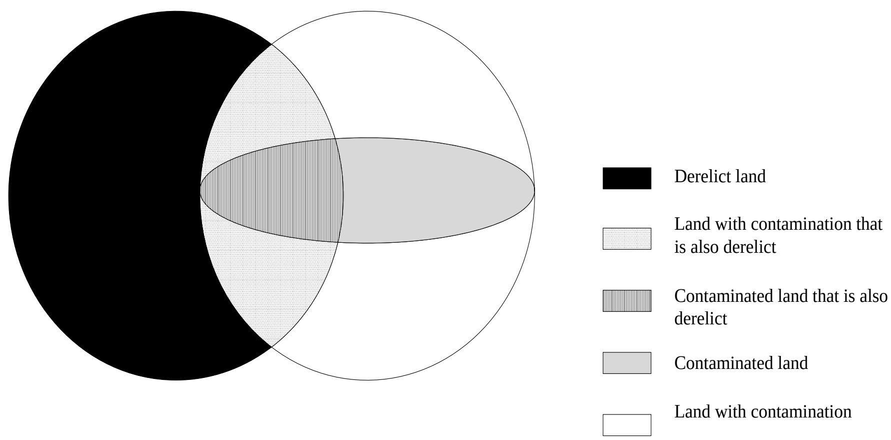

Figure 2.2 – Relationship between different definitions of contaminated land[5] (Pollard et al 2001)

Average national estimates of European Environment Agency (EEA) member countries show that on average approximately eight percent of the member country lands are contaminated and need to be remedied (EA 2007). However this figure needs to be taken with caution as there is no commonly accepted definition of contaminated land between member states (Carlon et al 2009), and different countries have different definitions,

[4] A Brownfield land is a previously developed land that could be vacant, derelict and / or contaminated.

[5] A derelict land is land that has become damaged from development and is beyond beneficial use without treatment. A PDL is that which is or has been occupied by certain permanent structure(s). Land affected by contamination is that which is known to contain harmful substances that do not meet the statutory requirements under the contaminated land regime.

legislations and priorities due to differences in extent and perception of the problem, and political and socio-economic backgrounds (Pollard et al 2001). However although the estimate is affected by lack of a common definition, it still correctly reflects the magnitude of the problem (Carlon et al 2009). Relatively little quantitative knowledge exist on the extent of the global scale of the problem, nevertheless there is little reason to believe that the situation is markedly different in other industrialised countries (Bridges et al 2006).

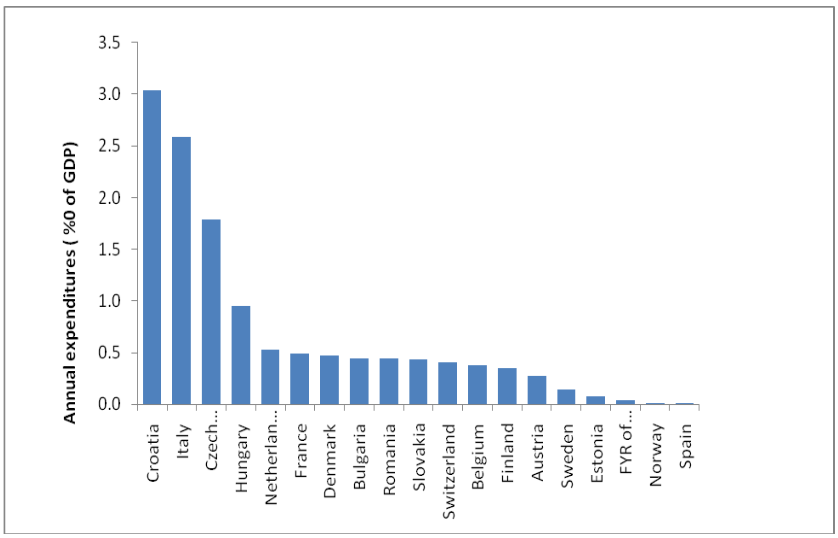

Figure 2.3 – Annual expenditure for the management of contaminated sites as $^{o}/_{oo}$ of GDP (EEA 2007)

The management of contaminated land is presently about two percent of the overall EEA member country management expenditure, with average annual expenditure of 12 EUR per capita (*Fig 2.3*). Although the cost of the same clean-up solution can vary by several orders of magnitude across member states, the cost per site is estimated to be on average between 19 500 and 73 500 EUR, with the total cost of remediated sites approximately 28 billion EUR (Carlon et al 2009). The EEA predict the number of identified contaminated sites to increase by 50 percent by the year 2025 due to increased level of awareness and commitment to the identification and characterisation of these lands (EEA 2007). To date over 80 000 sites have been cleaned up across the EEA member countries (*Fig 2.4*), and there still remains approximately 250 000 identified sites requiring clean up (EEA 2007).

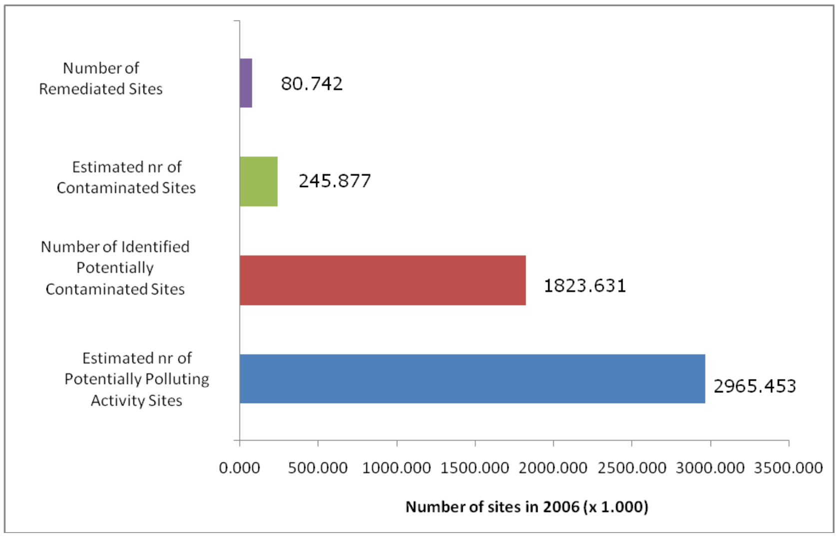

Figure 2.4 – Status in investigation and clean-up of contaminated sites in Europe (EEA 2007)

2.3 CONTAMINATED LAND PILICY: A UK PERSPECTIVE

Environmental policy in the UK has evolved substantially over the last decades both in domestic terms and as a response to European Community (EC) policy developments to ensure that it is not only relevant but proportional (Henton et al 1993). In the early days land contamination was merely costed for in the purchase of land for redevelopment (Young et al 1997). Both Government and public attitudes changed after a few high profile incidents like the Love Canal disaster (Beck 1979) and the Loscoe bungalow demolition from landfill gas (Young et al 1997). Contaminated land incidents then began to be perceived as very few and extremely severe incidents, with poorly understood but possibly disastrous consequences for human health (Vegter 2001). Policy became more conservative, aiming for maximum risk control (the principle of *multi-functionality*[6]).

As experience with the management of contaminated land has grown, the perception of the problem has changed significantly. Current policy regard land contamination as a widespread infrastructural problem with varying degrees of intensity and significance, and that returning all lands affected by contamination to pre-industrial standard is not only unnecessary, but technically and economically unfeasible (Ferguson 1998). As such current policy favours a risk based approach with clean-up standards based on site end-use. This focuses decision-making on areas where risks are unacceptable (Sheehan and Firth 2008). In the UK contaminated land policy is mostly restricted to the legacy of historic contamination. New contamination is considered separately under more stringent regulations since it could have been prevented (the *prevention principle*[7]) (Vegter 2001).

2.3.1 Contaminated land legislation

Contaminated land policy in the UK is set by the central Government but enforced and regulated by Local Authorities (LA). The contaminated land policy is closely associated both technically and legislatively with issues of redevelopment, groundwater pollution prevention and control, waste management and industrial site decommissioning (Pollard et

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[6] The multi functionality principle requires cleaning standards to be sufficient for any land end-use.

[7] The prevention principle requires the state of the environment should not get worse as a result of pollution that can be avoided. Further pollution of already polluted areas should be avoided. The principle also implies that accumulation of persistent substances in the environment should be stopped.

al 2001) and is dealt with through a number of regulations. These are[8]:

- The *Contaminated Land Regime* which is set out in Part IIA Environmental Protection Act 1990 (EPA). The regime sets out a joint regulatory role between LA and the EA to deal with the legacy of historical land contamination by identifying and remedying contaminated sites where there is an identifiable and unacceptable risk to human health or the wider environment.
- The *Planning System* which deals with existing contamination during redevelopment to ensure the land is fit for use. This is the primary means of dealing with contaminated land issues, as the majority of remediation is carried out during the redevelopment and regeneration cycle. The Part IIA definition of contaminated land still applies to management under the planning regime.
- The *Buildings Regulations 1991* applies to new developments to protect both the buildings and their future occupants from the effects of land contamination. The Part IIA definition of contaminated land also applies to management under the building regime. In the case of both new buildings and redevelopment, enforcement is by the Local Planning Authorities (LPAs), rather than by LAs.
- The *Water Resources Act 1991* is used for the prevention and removal of pollution from controlled waters. This is useful in situations where there is historic contamination and Part IIA does not apply, for example where the contamination is contained within the relevant water body or in cases of diffuse

[8] Part IIA EPA applies to Scotland and Northern Ireland too; however the principal regulator with regards Part IIA is the Scottish Environment Protection Agency (SEPA) and the Northern Ireland Environment Agency (NIEA) in Northern Ireland respectively, and equivalent agencies and consultants.

pollution where contaminant sources cannot be identified.

- The *EU Groundwater Directive* is used for the protection of groundwater resources from discharges and disposals of substances. This regulation is implemented in the UK through the *Groundwater Regulations 2009*.
- The *Environmental Permitting Regulations* (EPR) is useful in situations where contamination has resulted from land subject to waste management license, as Part IIA will not normally apply. There is also a duty of care under EPR for the safe disposal, transport and storage of waste-by products from remediation.

2.3.2 Regulatory roles and responsibilities

The risk based management policy is participatory with other Government agencies and departments, and other stakeholders (Pollard et al 2008), often involving statutory consultations and informal advice from various other Government agencies, departments, LA and organisations, with each playing a complimentary role: (DEFRA 2008):

- The *Environment Agency* (EA), which as the Government's principal adviser on the environment, is responsible for scientific and technical advice on contaminated land, for producing non-statutory technical guidance such as the CLEA[9] model (EA 2004*a*) and the Model Procedures for the Management of Contaminated Land (EA 2004*b*), *and* responsible for designated 'special sites'[10].
- The *Department of Environment, Food and Rural Affairs* (DEFRA) is responsible

[9] CLEA – *Contaminated Land Exposure Assessment*.

[10] A 'special site' is any contaminated land "which has been designated as such by virtue of section 78C(7) or 78D(6) of Part IIA EPA; and whose designation as such has not been terminated by the appropriate Agency under section 78Q(4) of Part IIA EPA".

for contaminated land legislation and all associated policy.

- The *Health Protection Agency* (HPA) is the principal scientific and technical adviser with regards to health effects from toxic substances.
- The *Food Standards Agency* (FSA) is the principal scientific and technical adviser with regards to food issues resulting for land contamination.
- *Local Authorities* (LA) are the principal regulators for contaminated land in their areas and are responsible for producing strategies for identification of contaminated land, for ensuring remediation takes place, for designation of special sites *and* for apportionment of liability.
- *Local Planning Authorities* (LPA) regulate the management of contaminated lands that is within the development or redevelopment cycle in their area.
- *Regional Development Agencies* (RDA) provide advice and guidance with regards to Brownfield regeneration and sustainable development.
- *Natural England* provides advice with regards to the impacts of land contamination on ecosystems and the natural environment.
- *English Heritage* provides advice with regards to impacts of land contamination on the historic environment, elements of cultural heritage and historic landscapes.
- Guidance on addressing impacts on biodiversity is jointly provided by the EA, Natural England, English Heritage and organisations like *the Royal Society for the Protection of Birds* (RSPB) and the *Centre for Ecology and Hydrology* (CEH).

2.3.3 Definition of contaminated land

Although the prevention of new contamination is of critical importance, the focus of Part

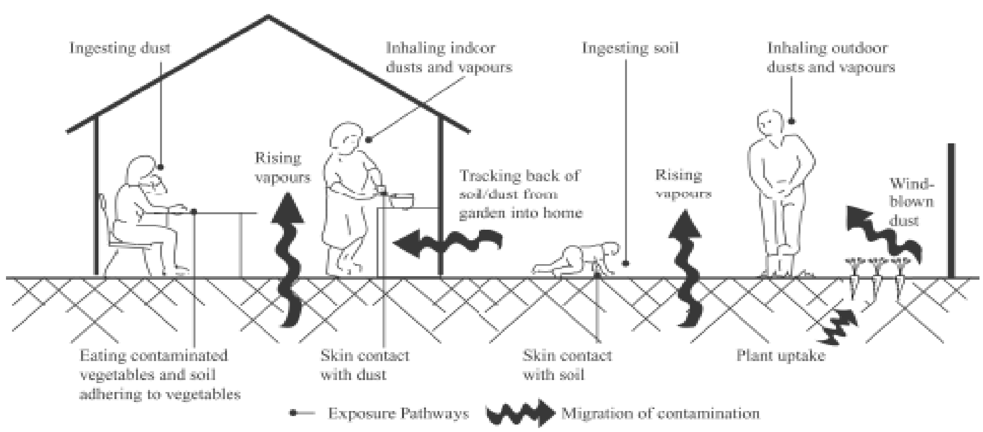
IIA legislation is on the substantial history and legacy of land contaminated, with new contamination dealt with separately (DEFRA 2006). Even with historic contamination Part IIA normally only applies when no better solution is available, such as in situations where redevelopment has already taken place without adequate treatment or in sites that require urgent action because the risks are too great to await redevelopment (DEFRA 2008). Contaminated land is statutorily defined in section 78A (2) Part IIA as:

> *"any land which appears to the local authority in whose area it is situated to be in such a condition, by reason of substances in, on or under the land, that –*
>
> *a. significant harm is being caused or there is a significant possibility of such harm being caused; or*
>
> *b. pollution of controlled waters is being, or is likely to be caused."*

Part IIA defines harm as "*harm to the health of living organisms or other interference with the ecological systems of which they form part, and in the case of man, includes harm to his property*". Property includes buildings, infrastructure and could be in other forms such as crops, livestock, domesticated animals and wild animals subject to shooting or fishing rights. The term '*significant*' is clarified in statutory guidance in relation to human health to include "*death, disease, serious injury, genetic mutation, birth defects or impairment of reproductive functions; with similar guidance in relation to property, the environment and non-toxic effects on humans*" (DEFRA 2008). Controlled waters are defined in the Water Resources Act 1991 to comprise estuaries, inland waters, groundwater and territorial waters. The pollution of controlled waters is defined in section 78A (9) of the same Act as:

"*the entry into controlled waters of any poisonous, noxious or polluting matter or any solid waste matter*."

In relation to pollution of controlled waters, section 78A of Part IIA stipulates "*controlled waters are 'affected by' contaminated land if (and only if) it appears to the enforcing authority*[11] *that the contaminated land in question is ... in such a condition, by reason of substances in, on or under the land, that pollution of those waters is being, or is likely to be caused*". The definition of groundwater in relation to Part IIA is clarified in the Water Act 2003 to include water below the saturation zone. The definition does not include all land where groundwater contamination is present, but such lands may be relevant under other regimes (DEFRA 2006). This ensures that the contaminated land regime deals effectively with situations where contaminating substances have left the surface of land, and are contained in underground strata, but have not yet fully entered the saturation zone.



Route		Route	
1	Ingestion of soil	6	Dermal contact with household dust
2	Ingestion of household dust	7	Inhalation of fugitive soil dust
3	Ingestion of contaminated vegetables	8	Inhalation of fugitive household dust
4	Ingestion of soil attached to vegetables	9	Inhalation of vapours outside
5	Dermal contact with soil	10	Inhalation of vapours inside

Figure 2.5 – An illustration of the potential human exposure pathways (DEFRA 2002)

[11] The EA in England and Wales, and SEPA in Scotland.

Key to the definition of contaminated land in Part IIA is the *pollutant-linkage* concept (*Fig 2.5*) where contaminant sources must be in concentrations sufficient enough to pose a Significant Possibility of Significant Harm (SPOSH) to human health and other receptors, such as other natural resources like air and water resources, ecosystems and habitats, and property, which includes crops, livestock and buildings. A sound pathway (*the linkage*) must exist between contaminant source(s) and receptor(s) for risk(s) to exist, and therefore the land to be contaminated under Part IIA legislation.

Other integral aspects of the risk based approach is the *fitness for use*[12] principle which recognises that different land uses require different soil quality, and for cleanup to therefore be proportional to site end-use, the protection of controlled waters, wider environmental and ecological protection and stewardship. The elements of the pollutant-linkage are defined in statutory guidance (Annex 3 of Part IIA EPA):

- *A contaminant source is a substance which is in, on or under the land and which has the potential to cause harm or to cause pollution of controlled waters*
- *A receptor is either:*
    - a. *Human beings, living organisms, group of living organisms, an ecological system or a piece of property ...; or*
    - b. *Controlled waters which are being, or could be, polluted by a contaminant.*
- *A pathway is one or more routes or means by, or through, which a receptor:*
    - a. *Is being exposed to, or affected by, a contaminant; or*

[12] Fitness for use principle aims at sufficiently reducing risks to human health and the environment as necessary to ensure the safe use or reuse of the land (CLARINET 2002*a*).

b. Could be so exposed or affected.

Risk to human health relates to the likelihood and magnitude of adverse effects from long term exposure (*direct or indirect*) of contaminants. Human health effects that are of concern include those related to chronic exposure including carcinogenic, mutagenic or toxic effects. Potential exposure pathways include direct dermal contact, inhalation, direct accidental ingestion of contaminants or indirectly through the food chain or from water consumption. The most commonly encountered exposure pathway is ingestion. Risks to ecological systems include direct adverse effects on soil organisms, plants and above ground wild life, and indirect effects on soil functions (Carlon et al 2009).

2.3.4 The Risk-based approach

The Part IIA definition uses a risk-based approach to determine when land is contaminated, and how stringent remediation must be to return it to beneficial use. In general, risk assessment evaluates the probability of occurrence of adverse effects. If the adverse effects have occurred, the consequences are known as damage (CLARINET 2002*a*). This is because in the vast majority of cases there is no appreciable risk, and a definition based on the mere presence of contaminants would cause large swathes of land to be caught unnecessarily. Taking contaminant concentrations in isolation of other risk factors is also not a good indicator of risk, as any given concentration may pose a markedly different level of risk depending on where it is and who/or what receptors may be affected (DEFRA 2008). The risk based approach therefore targets contaminated lands where there is a possibility of harm occurring, as low levels of both natural and anthropogenic contaminants are present in most soils and there is little land that has not been subject to some degree of contamination in the UK, albeit by long-range aerial depositions (Pollard et al 2001).

The risk based approach is often challenging however as it is often hard to estimate risks precisely because of the site-specific nature of risks, the diversity and heterogeneity of contaminants, and the variability in knowledge of the effects of contaminants on receptors. It is also often difficult to distinguish between SPOSH and non-SPOSH, as decisions on whether risks constitute SPOSH are taken on a case-by-case basis taking into account toxicological information, and site specific variabilities (DEFRA 2008). Despite these challenges however, the risk based approach is necessary in order to strike a balance between protecting human health and other resources, whilst minimising unnecessary socio-economic and environmental burdens (DEFRA 2008). Moreover not all of the impacts of land contamination are necessarily harmful. For example, an ecosystem could become dependent on some contamination conditions, and some contaminated sites could be part of a valued industrial heritage (CLARINTE 2002*a*).

The *precautionary principle* is applied in situations where: (*i*) SPOSH cannot be determined and there is a good reason to believe that it may occur; and (*ii*) the level of scientific uncertainty about the consequences or likelihood of the risk is such that best available scientific advice cannot assess the risk with sufficient confidence to inform decision-making (ILGRA 2002). However good reason still needs to be demonstrated by empirical evidence, expertise and/or sound theoretical explanation as to how SPOSH might occur. The purpose of the precautionary principle is to create an impetus for decision-making regardless of scientific uncertainty about risk, thereby preventing *paralysis by* *analysis*[13] by removing excuses for inaction on the grounds of scientific uncertainty.

2.3.5 Other policy drivers

Apart from human health and environmental protection, there are other key drivers for contaminated land management policy. The demand for housing and the associated development for infrastructure to support it are the main drivers for developing Greenfields (DEFRA 2009), although there are numerous derelict or contaminated sites (*Brownfield*) that could be redeveloped for this purpose[14]. The UK contaminated land policy addresses the problem from two main perspectives: (*i*) the protection of human health and the environment perspective; and (*ii*) the spatial planning perspective. A major policy trend is addressing these two perspectives simultaneously, with the development of integrated contaminated land management and redevelopment policies (Carlon et al 2009).

The conservation of land as a natural resource has led to policies that favour the redevelopment of Brownfield, which is seen as a sustainable land use strategy. As part of this, the UK Government has a Brownfield initiative, which encourages ways of responsibly dealing with increasing land use pressures by regenerating vacant or derelict PDLs in an effort to curb Greenfield consumption (*Fig 2.6*). This enables the recycling of more PDL than would otherwise be the case, increasing the ability to make beneficial use of the land (DEFRA 2006). The Governments target of 60 percent all new development to be on PDL under the Brownfield initiative has been met eight years ahead of target (*Fig 2.7*). The creation of Brownfield continues however, and some rehabilitation has not been

[13] Paralysis by analysis occurs when an outcome for a decision is never reached due to over analysing.
[14] The lack of a common definition of Brownfield has made quantifying the scale and extent of it difficult.

successful, leading to return of the land to derelict or underused state (CLARINET 2002*a*).

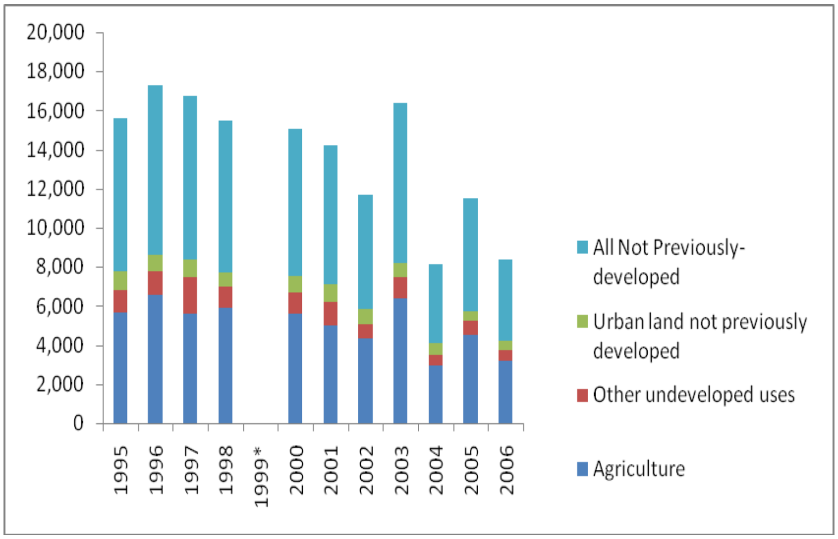


Figure 2.6 – Soil loss to development in England, 1994 to 2006[15] *(EA 2006)*

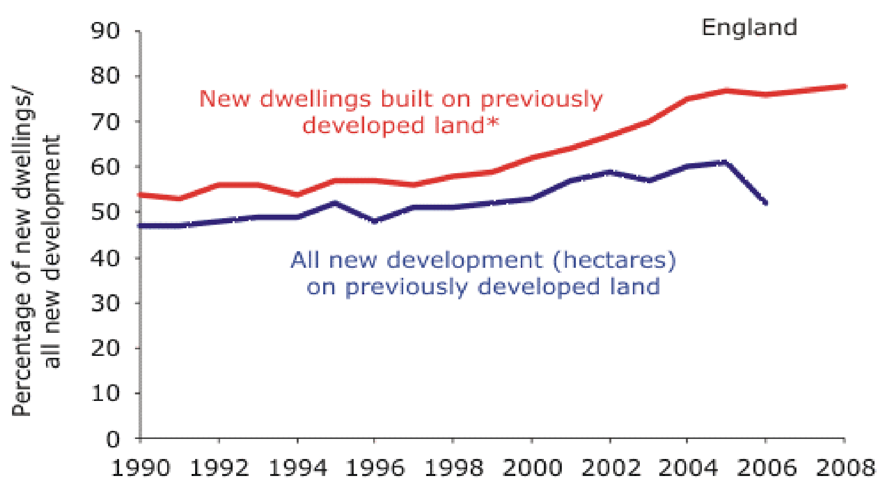


Figure 2.7 – Land recycling 1990-2008 (DEFRA 2009)[16]

A third emerging perspective is that of the sustainable management of contaminated land,

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[15] * 1999 data incomplete for absolute amounts.

[16] Include conversions. Up to 2002 conversion of existing buildings was estimated to add three percentage points, from 2003 the process of estimated has been elaborated.

in particular the need to consider the timing of any intervention and the future consequences of any particular solution in relation to at least economic, environmental and social criteria (*Fig 2.8*). The presence of extensive areas of contaminated or derelict land is one of the main challenges of sustainable land use, posing potential threats to achieving the Governments targets for sustainable development (CLARINET 2002*a*).

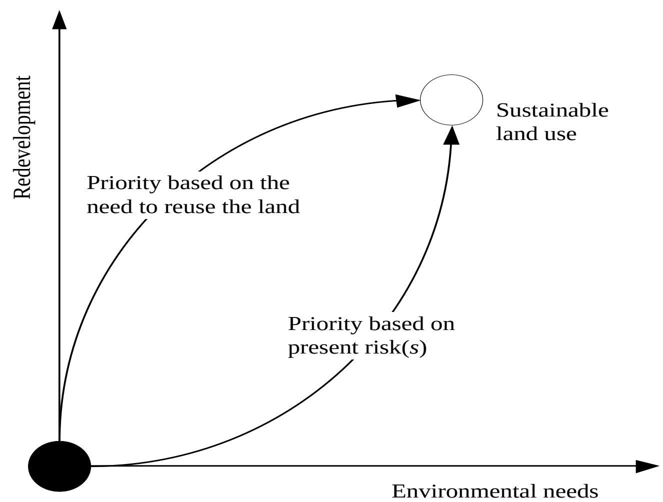

Figure 2.8 – The different trends in contaminated land policy (CLARINET 2002a)

The sustainable management of contaminated land is '*the practice of demonstrating, in terms of economic, environmental and social indicators, that an acceptable balance exists between the effects of undertaking the remediation activities and the benefits the same activities will deliver*'(Bardos et al 2009). Sustainable management of contaminated land therefore aims to find a positive overall solution that will achieve multiple gains and minimise regrettable losses (Gibsons et al 2005). This supports the Government's goal of sustainable development by helping the conservation of land as a valuable natural resource, reducing the pressure on Greenfield development and preventing the spread of pollution

(CLARINET 2002*a*). In the UK sustainability issues with respect to soil quality are addressed through a combination of policy, regulatory, voluntary and technological instruments, including (Pollard et al 2004):

i. Bringing land back into early beneficial use.
ii. Reducing pressure on Greenfield sites and the pollution of groundwater resources, thus conserving agricultural land and natural habitats.
iii. Adoption of a suitable-for-use approach towards land remediation.
iv. The efficient use of resources to tackle issues of highest risk at priority sites.
v. Prioritising remedial action so as to address the worst risks first in relation to the use of the land concerned.
vi. The application of sustainable remediation technologies that conserve land and resources.
vii. Development and maintenance of new partnerships and from key stakeholders with agreements on a common research and practice agenda.
viii. The consideration of types of sources of soil pollution over the long term.
ix. The development of monitoring systems that allows early detection of adverse soil, water and ecosystems changes.
x. The distribution of impacts from land contamination on communities.

2.4 CONTAMINATED LAND MANAGEMENT PROCESS

The effective management of contaminated land requires the integration of vast multi-disciplinary knowledge-bases into a coherent decision-making framework, taking into account the range of contexts in which decision have to be made, including complying with the relevant legislative framework, accounting for total operating costs and benefits, and addressing issues of environmental impacts, sustainability, protection of other resources, and importantly the prevention of further and/or future contamination (Bardos 2001). Land contamination also often involves different mediums (*soil, groundwater, surface water*). The management process therefore typically involves multi agency regulation and multidisciplinary expertise, with each discipline involved in interpreting discipline specific information for decision-making (Bardos et al 2001).

The contaminated land management process is complex and is typically undertaken using a phased approach (*Table 2.1*) with explicit considerations of risk at each phase of the decision-making process (Hester and Harrison 2001). With costs increasing at each stage of the management process, site investigation is a critical stage for decision-making, as it is the stage where the key decision is made as to whether the site is contaminated and if so whether the contamination is sufficient enough to warrant remediation. The cost of site assessment stage is reported to be is in most cases less than five percent of the overall project costs and in many cases may not even exceed one percent (Genske 2003). The importance of a thorough site investigation and assessment cannot be over emphasised as it could potentially prevent costly and unnecessary remedial action. Although each site is unique and requires a site specific solution, many of the key decisions are similar in structure. As a result many countries have developed generic national frameworks that integrate the key management decision-making processes (Bardos et al 2001).

Table 2.1 – Systematic approach to contaminated land management: from identification to characterisation, assessment and management

PHASE	ACTION	DESCRIPTION
1	Desk study	Information relevant to the whole management process is collected. Information collected may include geological maps and surveys, aerial photographs, historical information, land use, vegetation, water courses, oral evidence etc.
2	Preliminary site investigation	Site visit to collect site-specific information and confirm information collected from desk study.
3	Walk over survey	Extent and nature of contamination are identified, ground conditions and vegetation established in desk study are confirmed, and evidence of impact of contamination is established.
4	Chemical sampling and analysis	Information from all previous stages is assessed and evaluated, and the findings are used for designing a site-specific remediation strategy.
5	Remediation	Site is returned to beneficial use by either removing contaminants posing harm, treating the contaminants to reduce or eliminate harm or containing the contaminants by isolating them.
6	Site monitoring and aftercare	To ensure remediation is effective and management objectives have been fulfilled. Ongoing site monitoring may sometimes be necessary in cases where some level of contamination remain after remediation

The EA has developed a comprehensive technical framework for applying risk management to contaminated land (EA 2004*b*). This sets out a structured framework for assessment and decision-making within Government's policy and statutory requirements that could be adapted to apply to a range of management contexts. The framework uses a tiered risk based assessment approach, with each incremental tier involving increasing detail and complexity. These tiers are preliminary risk assessment, generic quantitative risk assessment and detailed quantitative risk assessment (DEFRA 2008).

- Preliminary (qualitative) risk assessment is undertaken to develop a site conceptual model based on information collected from desk study and site investigation phases. The conceptual model is used for identifying pollutant-linkages and is updated as more information becomes available. If a pollutant-linkage is found, then it may be necessary to proceed to the quantitative risk assessment or remedial action. If more than one linkage is found, it will need to be separately assessed and dealt with. Although professional judgement is used for assessment, decisions will still need to be justified both scientifically and technically.
- Generic quantitative risk assessment involves comparing contaminant concentrations with Generic Assessment Criteria (GAC) values. GAC values are generalised assessment criteria that are applicable to a wide range of soil types, site conditions (geology, hydrogeology, hydrology *etc*) and land use types. Although not legally binding, the EA and DEFRA Soil Guideline Values[17] (SGV) and the drinking water standards are used as GAC values for assessing risks to human health from soil contaminants and controlled waters respectively. Most practitioners still use withdrawn values such as the Inter-Departmental Committee on the Redevelopment of Contaminated Land (ICRCL) or the Dutch values rather than calculate SGV for unpublished contaminants, posing potential human health financial implications as they are not suitable for assessing the "significant possibility of significant harm to human health" in the context of the current

[17] Soil Guideline Values (SGVs) are DEFRA's scientifically-based GAC values for evaluating long-term risks to human health from chemical contaminants in soils.

contaminated land management regime (DEFRA 2002). The Land Quality Management Ltd (LQM) and the Chartered Institute of Environmental Health have published GAC values for extended range of contaminants that is in line with the current contaminated land statutory regime and associated policy. These values are a reliable alternative to calculating values for contaminants with no SGV.

- Detailed quantitative risk assessment is undertaken to determine SPOSH using site specific data. It may be used as the sole means of assessing risks or in situations where the outcomes of the preliminary or generic risk assessment are not adequate. A software model or support tool is normally used for estimating and evaluating risk. The key objective will be to establish a threshold limit for each contaminant of concern, a remedial target. This is the concentration limit below which the contaminant will not pose a potential risk to receptor(s).

Preliminary risk assessment often involves direct observation of the effects or consequences of the existence of a hazard, which could take the form of visible pollutants leaching into water or the observation of morbidity or death in livestock or crop. In many cases risk assessment is based on a prediction of the risk. This relies on a good understanding of site characteristics or modelling to estimate risks and how they might arise. The prediction of risk could introduce uncertainty in risk assessment however as:

- there may be incomplete understanding of risks, *or*
- modelling may produce imperfect representation of the real world, *and*
- sampling, analysis and other investigations may not provide an accurate reflection of the true or relevant characteristics of the site (EA 2004).

If the outcome of risk assessment requires further action, a risk management strategy is developed and implemented. In many cases the practical objective of risk management is to reduce risks rather than to eliminate them as total containment or removal of contaminants from complex and heterogeneous soil environment is rarely feasible (Bridges et al 2006). Risk management is therefore a much broader process than the selection of remediation technologies, and includes all the aspects of developing and implementing a sustainable solution (Vegter 2001). This involves remediation design and the appraisal and selection of appropriate remedial action(s), by ensuring that remediation is not only effective, sufficient and proportional to land end use, but carried out within the relevant legislative framework.

Remediation is the corrective action of cleaning up contaminated sites by eliminating or reducing the contamination to an acceptable level (Carlon et al 2009). The BATNEEC[18] principle is applied to ensure the Best Available Technology (BAT) is used, while considering costs, effectiveness and other secondary factors such as environmental impacts and sustainability of the remediation technology used. Remediation is often designed for either the total or part removal of the contaminant source(s), breaking or changing the pathway to receptors or relocating receptors. Remediation technologies broadly fall into one of these categories (*after* Janikowski et al 1998 and Carlon et al 2009):

- Excavation of the contaminated soil for disposal elsewhere, followed where necessary by replacement with clean material. *Ex-situ*[19] technologies are applied to excavated soil and/ or extracted groundwater.
- Engineering systems including isolation or containment of the contaminated soil by

[18] BATNEEC – Best Available Technology Not Entailing Excessive Costs.
[19] Ex-situ remediation is carried out above ground, and could be off-site.

covering it with a suitable thickness of clean, inert fill or hard cover. These could involve both *In-situ*[20] and *Ex-situ* technologies.

- Treatment based approaches for destroying, removing, cleaning or immobilising contaminants, and could include chemical, biological or physical processes.
- Site rehabilitation measures such as growing grass cover to bring back some utility to sites that cannot be treated or contained due to technical or economic reasons.
- Mixing the contaminated material with clean soil or sub-soil in order to reduce the maximum concentrations of contaminants to below the threshold trigger values.

The remediation technologies used for remedying contaminated sites strongly depend on several factors, including the nature, concentrations and physical states of pollutants present, the type of soil and specific aspects of the site itself (Rulkens et al 1993). Remediation also requires consideration of other factors, including balancing inevitable trade-offs between economic, environmental, social and technical criteria with respect to set management objectives and regulatory requirements. Increasingly remediation strategies are moving from technology based approaches to integrated treatment systems (treatment trains) that focus on land use management and the use of emerging technologies such as natural attenuation and phytoremediation (James and Kovalick 2002). Treatment trains are necessary in order to provide lower cost and more effective remediation solutions for complex sites (James and Kovalick 2002). Remediation is also increasingly focussing more on in-situ, area wide approaches rather than the traditional ex-situ, site specific approaches. Emerging technologies usually require much longer clean up periods however, and need to

[20] In-situ remediation is carried out in place, i.e. without removing the contaminated media.

be balanced with other management objectives.

Experience has shown there is no universal practical solution, with each solution having its advantages and disadvantages depending of site conditions, the nature and extent of the contamination, regulatory requirements and remediation objectives (CLARINET 2002*a*). Doubts still exist over the efficacy of many remediation technologies (LHC 1994), and the question remains as to whether remediation in itself is sustainable (CL:AIRE 2007). Each of the currently available remediation technology has significant drawbacks either economically, environmentally, socially or technically. Additionally, the biological functioning of the soil is often impaired during the cleaning process because of destruction of the microbial system and soil structure (Janikowski et al 1998).

2.5 CONCLUSION

Land contamination is a major environmental and infrastructural problem in industrial countries, with potential detrimental effects on human health, valuable water resources, sensitive ecological systems, property and infrastructure. The effective management of contaminated land typically involves multi-agency regulation and multidisciplinary expertise. This requires the integration of vast multidisciplinary knowledge-bases into a coherent decision-making framework, within current regulatory framework(s).

In considering the best course of action several factors must be taken into account such as site specific constraints, total operational costs and benefits, engineering feasibility, potential environmental impacts, sustainability and site monitoring and aftercare. Increasingly the goal of remediation is on the sustainable management of the contamination involving either full or partial treatment, isolation, or removal of contaminants on site. A solution with long-term aftercare may not be cost effective, and therefore economically unsustainable and possibly prohibitive, especially with cost often being the overriding factor in decision-making (Pollard et al 2004). It is also possible that a solution that appears suitable and is sufficient and proportional to land end-use may not be feasible technically or economically. A solution that takes short-term view of cost in lieu of longer-term financial and economic implications could result in a negative relationship between remediation costs and that of monitoring and aftercare (Pollard et al 2001).

Sustainable management involves balancing inevitable trade-offs between competing economic, environmental and social criteria, with ideal (sustainable) solutions aiming to minimise total operational costs, minimise environmental impacts and maximise social benefits. The ideal is rarely achieved on the basis on scientific evidence alone, and increasingly decision-making techniques and Decision Analysis (DA) methods are used to support with balancing the inevitable trade-offs between decision criteria.

3 CONTAMINATED LAND DECISION-MAKING AND DECISION ANALYSIS METHODS

3.1 INTRODUCTION

From the generalisation of contaminated land management process in the previous chapter, the decision-making process may be perceived to be quite structured and unproblematic. Contaminated land management is quiet complex in practice however, as each site is unique and requires site specific assessment and decision-making, with considerable administrative, financial, scientific and technical efforts (Gatchett et al 2007). Given the site specific extent and nature of contamination and the complexity of soil environments, site specific uncertainties will always be present (Hestor and Harrison 1997). Contaminated land management decision-making can therefore be quiet complex and is typically undertaken under conditions of risk and uncertainty, often resulting from (Vegter 2001):

- heterogeneity and complexity of soil environments,
- heterogeneity of the nature and extent of contamination sources,
- complexity of contaminant fate and transport,
- incomplete/inaccurate results from site monitoring or modelling,
- incomplete or incorrect understanding of risk(s),
- inaccurate reflection of true or relevant site characteristics,
- decision maker(s) assumptions,
- the use of both quantitative and qualitative information, *and/or*

- incomplete knowledge as full information is rarely fully attainable or available.

3.2 CONTAMINATED LAND DECISION-MAKING PROCESS

Due to the multidisciplinary and site specific nature of contaminated land management process, the decision-making process is typically characterised by complex trade-offs between competing and often conflicting economic, environmental and socio-political criteria which cannot all be fully satisfied in most cases (Schmoldt et al 2001). The same site may also consist of different environmental media (*e.g.* soil, groundwater and/or ecological systems) and therefore pose different types of risks, which may need to be dealt with under multiple regulatory frameworks, often involving statutory consultations with other Government departments and agencies and/or other stakeholder(s). There is also a range of contexts in which decisions have to be made, including relevant regulatory framework(s), operational costs and benefits, environmental impacts, sustainability, suitability and proportionality of remediation techniques (Bardos et al 2001).

Effective management of contaminated land therefore requires a good understanding and integration of vast multidisciplinary knowledge-bases for decision-making. Although each site is unique and requires a site specific solution, many of the key decisions are similar in structure, and as a result many countries have developed generic management frameworks that can be adapted to different management scenarios. Different decision-making techniques have been used for integrating multidisciplinary information into usable knowledge for decision support on the best course of action (Bardos et al 2001). A generalised decision-making process for contaminated land management begins with problem definition using site specific information and knowledge about the extent and nature of the contamination to define management objectives, within set regulatory constraints (*Fig 3.1*). For a single site the objective may be to remediate to a level suitable for residential or commercial land use, and for a series of contaminated sites, the objective may be to prioritise which site to remediate first to minimise risks whilst maximising amount of land available for use (Bardos et al 2001).

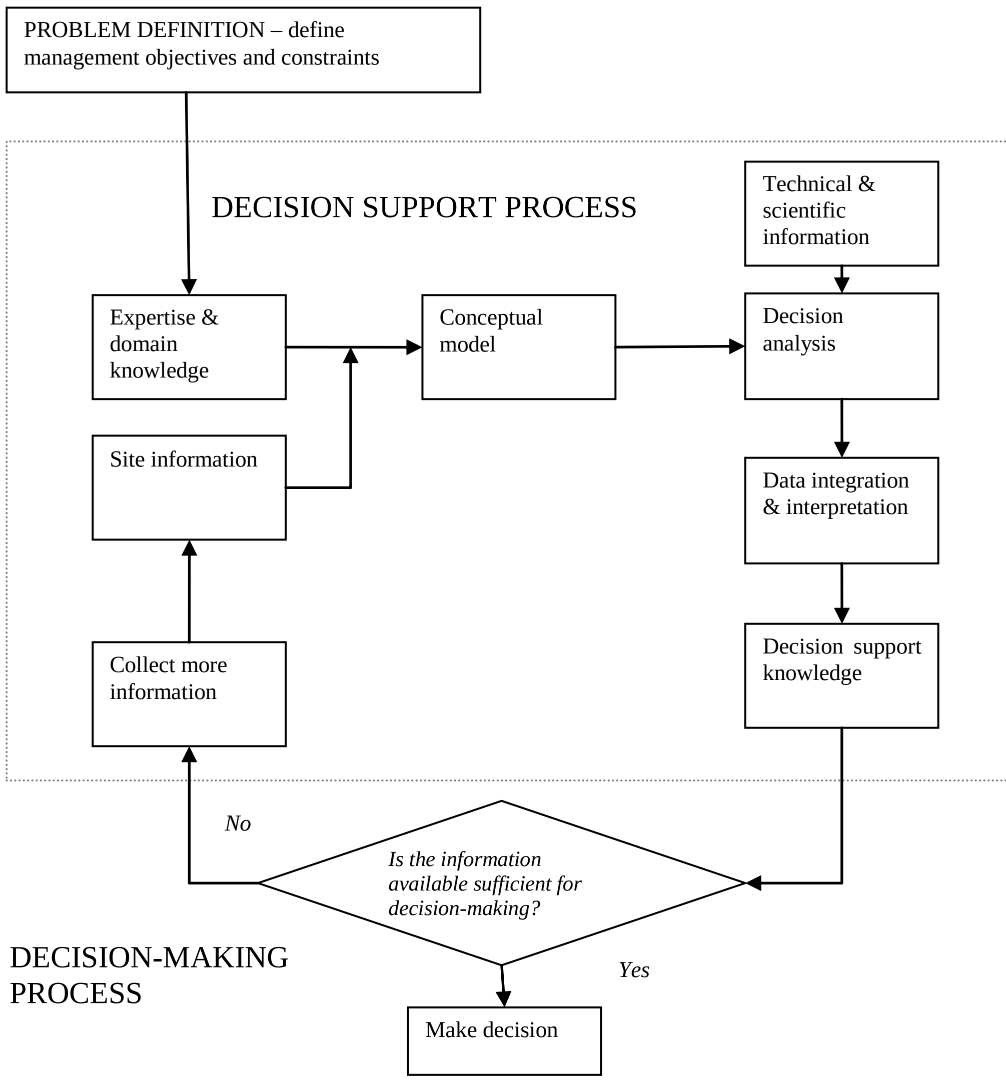

Figure 3.1 – Key steps in the contaminated land management decision support process (after Bardos et al 2001)

The decision support process assists with the identification of the best course of action, within set management constraints. This makes use of professional expertise, domain knowledge and site-specific information (*e.g. the nature and extent of the contamination, pollutant-linkages, land end use, etc*) and other site-specific data (*e.g. soil properties, hydrogeology, hydrology, etc*) to develop a conceptual model of site behaviour. The conceptual model is used with technical and scientific information for Decision Analysis (DA), which is a logical approach to decision-making under conditions of risk and/or uncertainty. The DA process begins with the identification of decision alternatives and decision criteria to be used for decision-making, which need to be concise, non-redundant and measurable either quantitatively or qualitatively (Giove et al 2009).

A decision-making tool is then selected for evaluating decision alternatives against criteria. This could be for different aspects of management decision support such as with risk assessment or risk management, comparison of remediation techniques or sustainability appraisal (*Table 3.1*). Different decision-making techniques and methods have been used for decision support. These techniques could be for decision-making based on a single management criterion, *e.g*. analysing management costs and benefits or assessing remediation cost-effectiveness. Multi Criteria Decision Analysis (MCDA) methods are used to support decision-making involving more than one decision criteria by balancing inevitable competing trade-offs amongst the criteria. The solutions from the analysis are finally integrated and then interpreted, into usable knowledge to support decision-making, which should be sufficient, transparent and communicable to all decision maker(s). The decision support knowledge should be proportional to the decision context, as insufficient or excessive information could impact the helpfulness of the knowledge for decision support, with possible impacts on quality of decision support provided (Bardos et al 2001).

Table 3.1 – Contaminated land management decision-making issues (after Bardos et al 2001)

MANAGEMENT CONTEXT	EXAMPLES OF ISSUES TO ADDRESS
Site characterisation	What is the nature and extent of contamination?
	Are there any pollutant-linkages?
	Are the contaminants in sufficient concentration to cause harm to any receptors (SPOSH)?
Regulatory framework	Is the site historically or newly contaminated?
	Is the site within a building or redevelopment cycle?
	Should the site be designated a special site?
	Who is the principal regulator?
	Is any statutory consultation necessary with other Government agencies or departments?
	Will land use change be required?
	Are there any other stakeholders like land owners or interest groups?
Risk assessment and management	What is the intended use of the land?
	Are there any site or engineering constraints?
	Slow extensive remediation *vs*. fast intensive approaches?
	What are the total operating costs?
	Will the cost of the remediation outweigh its benefits?
Remediation	What is the *Best Available Technology* (BAT)?
	Will remediation have any impact on local ecology and the natural environment?
	Will remediation be sustainable?
Monitoring and aftercare	Will there be long term aftercare after remediation?

3.3 CONTAMINATED LAND DECISION ANALYSIS

Decision Analysis (DA) is the practical application of normative decision theory to complex real world decision problems. Decision theory is a discipline that models human decision-making processes. There are two distinct decision-making theories: (*i*) normative (also *prescriptive*); and (*ii*) descriptive (Howard 1968). Normative decision-making deals with rational choice, and involves developing normative models based on decision maker's assumptions (*axioms*) and preferences. A further distinction is sometimes made between normative and prescriptive decision-making, because although prescriptive decision-making uses normative models, it takes into account limitations of human judgment and of the practical problems of developing rational models for complex real world decision problems. The second type of decision-making is descriptive decision-making, which deals with how real decisions are made, and involves developing models of actual human behavior (Edwards et al 2007). The distinction between normative and descriptive decision-making is therefore in principle very simple – normative decision-making is concerned with how decisions should be made, and descriptive decision-making is concerned with how decisions are actually made (Hansson 2005).

The contaminated land DA process is descriptive, as it does not predict or describe how decisions will be made, but rather facilitate better decisions than would otherwise be possible (Schmoldt et al 2001). The objective of contaminated land decision-making is to provide information to decision maker(s) on the decision situation, its constraints, alternative courses of action and their consequences (Schmoldt et al 2001). The type of knowledge used for supporting decision-making determines the decision situation, which could be made under conditions of: (*i*) certainty; (*ii*) risk; *or* (*iii*) uncertainty (Malczewski

1999). Decisions under certainty are made when each alternative leads to one outcome, and the consequence of each outcome is known. Decisions under risk are made when the outcomes will have one of several possible consequences, and the probability of occurrence for each consequence is known with full or partial certainty. Decisions under uncertainty are made when alternatives are known, but the consequences and the probabilities of the outcome are partly known, completely unknown, or in some cases not even defined. Decision constraints are set by site conditions, the nature and extent of contamination, land end use, the decision context and the policy requirements that must be satisfied (Bridges et al 2006). Contaminated land management decision-making is typically made under risk and uncertainty. Whilst the risk-based management approach is largely a scientific process, decisions must ideally balance the outcomes of the science with other criteria in order to fully satisfy the management objective (Bridges et al 2006).

Different decision-making techniques have been widely used to support contaminated land decision-making (*Table 3.3*). These techniques are suitable for supporting single criterion decision-making such as costing or environmental impact. The most commonly used technique for contaminated land management is Cost Benefit Analysis (CBA) because cost is often the overriding decision factor in contaminated land management (Pollard et al 2001). Contaminated land management decision-making characteristically involves multiple criteria however, and should ideally consider at minimum three criteria (economic, environmental and social) for decision-making (Vegter 2001). Using only one decision criterion, even one at the highest abstraction level can therefore not be regarded as a sufficient management approach (Janikowski et al 1998). An ideal management outcome is considered as one that effectively balances inevitable trade-offs between decision criteria to minimise costs and risks, whilst at the same time maximising benefits (Linkov et al 2006*b*). Such ideal rarely exists however and as such formalised and structured DA methods like MCDA are used to provide alternative means of evaluating these complex decision problems involving multiple criteria, and are used for balancing trade-offs between the competing and often conflicting management criteria (Linkov et al 2006*a*).

3.4 MULTICRITERIA DECISION ANALYSIS APPLICATIONS

Decision-making techniques offer little guidance on how to integrate or judge the relative importance of alternative decision outcomes, and do not provide structured means of arriving at optimal outcomes, or provide a means for incorporating different types or scales of information, or the multiple stakeholder preferences that must typically be brought to bear in management decision-making (Kiker et al 2005, Linkov et al 2006*a*). It is widely accepted that taking management decisions in isolation is no longer sufficient and there is a need for a robust and integrated decision-making framework (Pollard et al 2004). MCDA has been found to be especially useful to environmental management decision problems which require balancing scientific findings with multifaceted, value laden input from many different stakeholders with different priorities and objectives (Giove et al 2009).

MCDA methods provide scientifically sound decision-making framework for decision problems where criteria such as costs, environmental impacts, safety, and risk cannot be easily condensed into simple monetary expressions (Linkov et al 2006*b*). MCDA methods have several other advantages, such as providing consistency, documentation, rationality and transparency to the decision-making process, thereby increasing confidence in the decision outcome. MCDA applications for risk based management generally use different types of information for decision making, ranging from extremely qualitative to extremely quantitative, including: (*i*) the results from site monitoring or modelling; (*ii*) risk analysis; (*iii*) CBA; *and* (*iv*) stakeholder preferences (Linkov et al 2006*a*). MCDA allows for the integration of both quantitative and qualitative information for decision-making.

MCDA algorithms are designed to synthesise a wide variety of information and raise awareness of the trade-offs that must be made between competing management objectives, and provide a systematic approach for integrating risk levels, uncertainty and technical valuations (Giove et al 2009). The MCDA process is a structured approach of choosing from amongst multiple decision outcomes, and has been shown to offer significant improvements in contaminated land management decision-making (Bridges et al 2006, Linkov et al 2006*b*). MCDA decision problems commonly include the same structural components (Giove et al 2006):

- an objective or target function to be optimised;
- a set A of alternatives, and in the finite case: $A = \{ \alpha_j : j = 1,2,\ldots,m\}$;
- a finite set of criteria $C = \{\alpha_i : i = 1,2,..n \}$ for evaluating decision alternatives;
- the decision maker(s);
- the decision maker(s) preferences; *and*
- an algorithmic tool for optimising the *objective function*, with respect to all of the components above.

The MCDA decision model can be generally described as $M_p = (A, D, C, \Omega, R)$ - where A is the finite set of alternatives, D is the set of consequences, C the criteria model, Ω the imperfect knowledge and R the aggregation procedure (Figueira 2006). The MCDA process is the same as with DA process, and begins with defining management objectives, and indentifying decision alternatives and criteria for evaluating the alternatives. The identification of decision alternatives and criteria is generally quiet subjective, and the least technical part of the MCDA process (Henig and Buchanan 1996). A set of alternatives are judged against a set of criteria by assigning values to each criterion for each action (Janikowski 1998). A rating matrix with a set of alternatives $(A1, A2, \ldots Ai)$ along one axis and a set of criteria $(C1, C2, \ldots Cj)$ along the other is constructed (*Fig 3.2*).

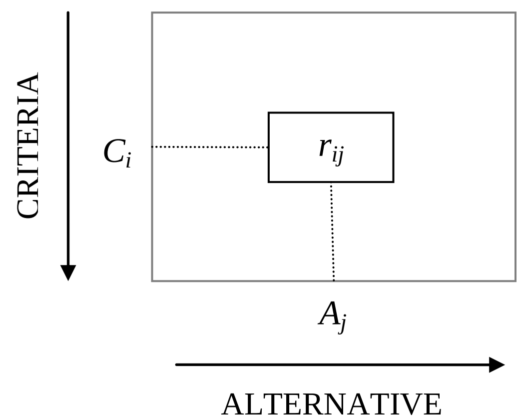

Figure 3.2 – Rating matrix (Janikowski 1998)

Further analysis is undertaken in MCDA, beginning with eliciting the decision maker(s) preferences by assigning weights according to their relative importance with respect to the management objectives (*Fig 3.3*). The second phase permits the elaboration of the hierarchy between the different alternatives and consequently the final choice of the best alternative. The final stage involves sensitivity analysis to evaluate the degree to which variations in inputs can influence the final result (Giove et al 2009). Depending on the decision problem, different types of MCDA methods are used for evaluating alternatives and criteria, including Multi Attribute Utility Theory (MAUT), the Analytical Hierarchy Process (AHP) and outranking techniques. All the methods are equally theoretically valid, and no better or worse method exist. The effective application of any evaluation process depends on the decision problem itself and how it is formulated.

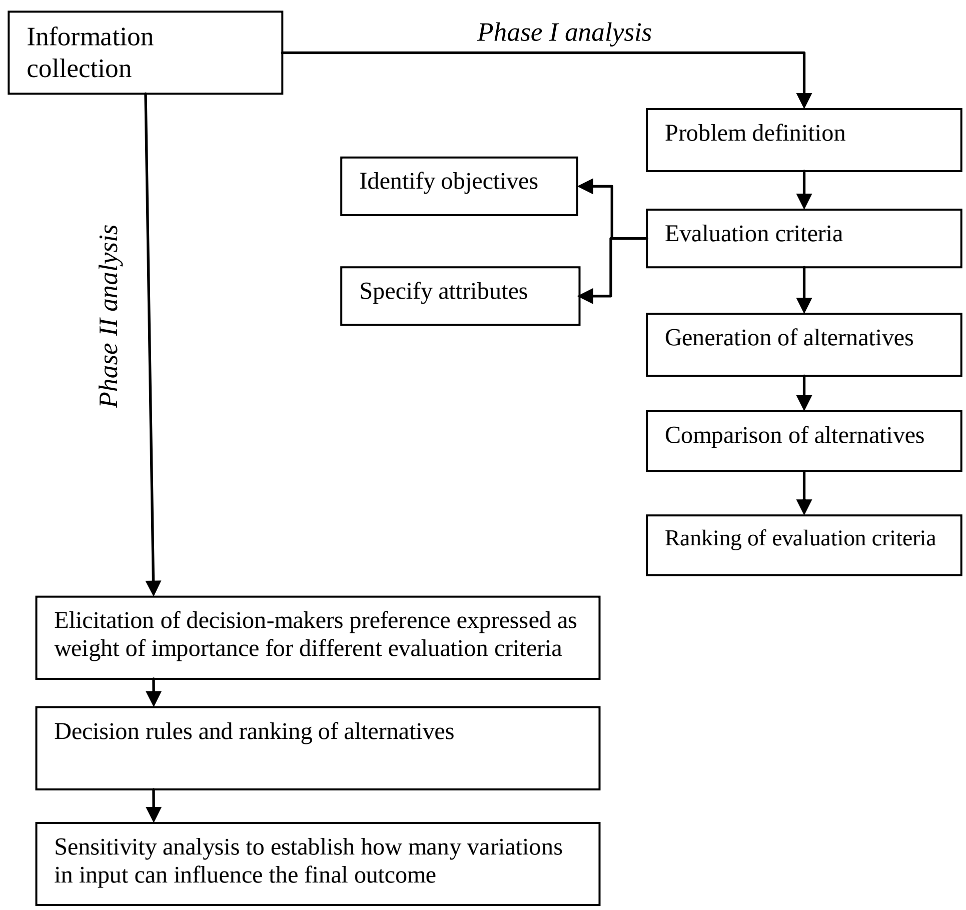

Figure 3.3 – Decision Analysis process for decisions with single or multiple objectives (Giove et al 2001)

Different terms are used for MCDA in the literature often interchangeably. It is therefore necessary to clarify the technical classification and usage of terms. MCDA is a broad family of formalised and structured DA methods that are used for supporting complex decision-making involving multiple criteria, which are classified according to different factors, including: (*i*) number of decision alternatives; (*ii*) decision-making under condition of certainty or uncertainty; *and* (*iii*) solution method used (Malczewski 1999, Giove et al 2009***)***. The first classification is based on the number of decision alternatives – Multi Attribute Decision Analysis (MADA) and Multi Objective Decision Analysis (MODA) methods (*Table 3.2*). MADA methods are used for decision problems with simple and discrete number of decision alternatives, while MODA is used for problems with large and continuous number of alternatives (Cho et al 2003, Zanakis et al 1998, Giove et al 2009).

Table 3.2 – Comparison of multi objective decision analysis and multi attribute decision analysis methods (Malczewski 1999)

	MODA	MADA
Criteria	Objectives	Attributes
Objectives	Explicit	Implicit
Attributes	Implicit	Explicit
Constraints	Explicit	Implicit
Alternatives defined	Implicit	Explicit
Number of alternatives	Infinite (large)	Finite (small)
Decision-makers control	Significant	Limited
Decision modelling paradigm	Process oriented	Outcome oriented
Application	Design / search	Evaluation / choice

The second classification is based on decision-making conditions, which could be either under certainty or uncertainty. In decision-making under certainty, the decision maker(s) has full knowledge of the decision situation, as well as exhaustive information about the decision-making process. In decision-making under uncertainty, the decision maker(s) has incomplete knowledge or understanding of either the decision situation or from incomplete understanding of decision alternatives or criteria importance. Both MADA and MODA methods can be used for decision-making involving individual or group decision makers (Malczewski 1999, Giove et al 2009).

The third classification is based on the solution method used. MODA uses Multi Objective Optimisation (MOO) for optimising trade-offs between competing objectives. Optimum in MODA is commonly described as a *Pareto optimal front*, where the outcome is considered optimal if no objective could become better off without another being worse off. MADA solutions could be either compensatory or non-compensatory. In compensatory methods, explicit tradeoffs are allowed between attributes (criteria), where poorly scoring criteria can be compensated for by higher scoring criteria. Compensatory MADA models are based mainly on utilitarian based methods like the MAUT and the AHP; and non-compensatory methods are based on outranking techniques.

In outranking, criteria weights are based on *coefficients of importance*, where a bad value on a criterion cannot be offset by good values on other criteria. MADA problems are assumed to have predetermined and limited number of alternatives, and therefore solving MADA problem is a selection process as opposed to MODA, which is a design process (Cho et al 2003, Malczewski 1999). Contaminated land management decision problems are characteristically MADA. And although MADA is technically a sub-type of MCDA, the term MCDA is often used interchangeably with MADA in the contaminated land literature. To that effect MCDA is used throughout the rest of the thesis to mean MADA.

Table 3.3 – Decision-making techniques and their application to contaminated land management

TECHNIQUE	DESCRIPTION	LIMITATIONS	EXAMPLE OF APPLICATION
Cost Benefit Analysis (CBA)	CBA is used for economic analysis of decision alternatives, and is used for assessing total operational costs and expected benefits of a project.	It is not always possible to assign monetary values to decision variables, and therefore it is often difficult to accurately and adequately estimate project costs and benefits. Moreover it is not always possible to determine whether the least costly alternative is the most beneficial in the long-term.	CBA has been used for analysing total management operational costs and benefits (Day et al 1997), for comparing the costs of alternative remediation techniques (Kavanaugh 1996) and for environmental policy making (Hanley 2001).
Cost-Effectiveness Analysis (CEA)	CEA is used for comparing relative effectiveness and benefits of alternative courses of action, with respect to their cost. CEA is used when measurement of benefits (*e.g*. in CBA) in monetary terms is not possible.	Like '*benefits*' in CBA, it is often hard if not impossible to assign monetary values to '*effectiveness*'. It is therefore difficult to adequately quantify effectiveness. CEA tends to focus on the direct results that occur over the short to medium term, and is only effective when full knowledge is available or attainable.	CEA has been used for assessing the effectiveness of ecosystem restoration (Macmillan et al 1998) *and* also has been used for assessing the effectiveness of remediation strategies (Day et al 1997).
Strengths, Weaknesses, Opportunities & Threats (SWOT) analysis	SWOT analysis is used for maximising project strengths and minimising its weaknesses with respect to external opportunities and threats that are necessary for satisfying management objectives.	SWOT analysis does not provide means of identifying SWOT elements, or any means of critically evaluating what constitute project SWOT.	SWOT analysis has been used for financial appraisal of contaminated land (Geneletti et al 2007).

Comparative Risk Assessment (CRA)	CRA is used for comparing risks, by distinguishing actual risks from potential exposure. CRA is a useful tool for rapid risk assessment from available information (WBG 1998). CRA can help provide transparency, scope and context for risk assessment process.	CRA can introduce bias in decision-makers preferences. CRA has often been criticised for being ambiguous, as risks could be so heterogeneously qualitative that a meaningful comparison cannot be made (Schutz 2007).	CRA has been mostly applied within the realm of policy analysis (Linkov et al 2006a), and has been used for analysing different risks from alternative remedial actions (Suter et al 2006).
Environmental Risk Assessment (ERA)	ERA is used for analysing potential hazards to receptors. ERA is good for highlighting and prioritising management needs, either qualitatively or quantitatively and has no requirements to produce monetary evaluations.	The ERA process is time consuming and costly, and sufficient data is not always attainable or available.	ERA has been widely used for ecological/human health risk assessment, *e.g*. for dealing with uncertainties in exposure assessments, and for the selection of appropriate remedial technologies (Pollard et al 2004).
Environmental Impact Assessment (EIA)	EIA is used for assessing the likely environmental impacts of projects. EIA consists of exploring alternative futures in a way that provide information of utility to decision makers (Duinker et al 2007).	EIAs are very time consuming and costly, especially in the data collection stage. Its findings are also sometimes found to be poor or biased if not carried out properly.	EIA has been used for assessing the likely environmental consequences of decision alternatives (Duinker and Greig 2007).
Social Impact Assessment (SIA)	SIA is used for assessing the social consequences (*e.g*. impacts on local populations) of projects. SIA is useful technique for involving stakeholders in the decision-making process (Pollard et al 2004).	Costly and time consuming, and like in EIA, '*impacts*' may be hard or impossible to quantify.	SIA has been used for identifying key benefits and constraints on remediation (Pollard et al 2007) – *e.g*. in assessing potential social impacts of projects on people (*e.g*. BP 2002).

Life Cycle Assessment (LCA)	LCA is used for evaluating project resource requirements and environmental impacts of products and services for project life cycle (Seppälä et al 2001). The LCA '*cradle-to-grave*' approach allows for comprehensive analysis of entire project life cycle.	There are many approaches and tools for LCA and the lack of uniformity between these makes the interpretation of results hard. LCA has also been criticised for being highly subjective, and sometimes of producing ambiguous results.	LCA is useful for providing information on the environmental impacts of remediation (Miettinen and Hamalainen 1997).
Scenario analysis	Scenario analysis is used for analysing different possible future scenarios (*forecasts*) and evaluating the possible outcomes of these by assigning probabilities and assessing the implications of these outcomes. These forecasts are useful in evaluating the long-term effectiveness of management strategies.	Scenario analysis is valuable in understanding implications of multiple possible forecasts; however scenario analysis should not be used as an end in itself, but as part of the decision-making process (Holroyd et al 2007).	By working with scenarios of quite different futures, focus is shifted from estimating what is most likely to occur – *i.e.* predictions, to the consequences of the different predictions and responses for these (Duinker and Greig 2007).
Adaptive management	Adaptive management is used for decisions which are able to adapt as new information is obtained (Cannon 2007).	Adaptive management is likely to be costly and slow in many situations (Lee 1999).	Adaptive management has been primarily limited to a few large-scale projects in long-term natural resource management, where uncertainty is so overwhelming that optimization is not possible (Linkov et al 2006a).

3.5 MULTICRITERIA DECISION ANALYSIS APPLICATIONS TO CONTAMINATED LAND MANAGEMENT DEICISON-MAKING

MCDA has been widely used to support different levels of management decision-making and many MCDA applications have been reported for contaminated land management (*e.g*. Kiker et al 2005, Linkov et al 2004, Linkov et al 2006*a*, Linkov et al 2006*b*, Pollard et al 2004 *etc*). These applications have been found to be beneficial to contaminated land management as they offer significant improvement in the decision-making process by providing structure, consistency, transparency, documentation and justification to the decision-making process (Bardos et al 2001, Bridges et al 2006). Different MCDA methods have been used, with some methods ranking options, some identifying a single optimal alternative, some providing a partial/incomplete ranking, and others differentiate between acceptable and unacceptable alternatives (Linkov et al 2006a). The methods have varying degrees of complexity, and can be broadly categorised as: (*i*) utilitarian methods of MAUT and derivatives; (*ii*) outranking techniques; *and* (*iii*) the AHP.

3.5.1 Utilitarian methods

The most commonly known MCDA method is the Multi Attribute Utility Theory (MAUT), which is based on the expected utility theory of von Neuman and Morgenstern (1953). MAUT is an objective and structured method of maximising a decision maker(s) utility (Keeney and Raiffa 1976). MAUT assumes that the decision maker is rational, has full knowledge and is attempting to maximise an utility (the decision maker's preference) from multiple alternatives (Cho et al 2003). An *utility function* is used to find simple expression for the net benefits of a decision outcome (Linkov et al 2006). The fundamental objective of MAUT is to model decision maker's preference using utility function $u(x)$ to express the utility of each criterion $x_1, x_2, ..., x_n$ in a common scale. A single attribute utility function for each criterion is derived as $u_1(x_1), u_2(x_2), u_3(x_3), ..., u_n(x_n)$. A multi attribute utility function is derived by: (*i*) assigning an *importance weight* $w_1, w_2, ..., w_n$ to each single attribute utility function; (*ii*) multiplying each single attribute utility function by an importance weight reflecting the decision makers preferences (*Eq. 1*) *and* (*iii*) summing the weighed single attribute utility functions (*Eq. 2*). The importance weight of all criteria should sum to a whole – 1 or 100%.

$$u(x_1, x_2, ..., x_n) = w_1 u_1(x_1) + w_2 u_2(x_2) + ... + w_n u_n(x_n) \qquad (\textit{Equation 1})$$

$$u(x_1, ..., x_n) = \sum_{i=1}^{n} w_i u_i(x_i) \qquad (\textit{Equation 2})$$

Utilities are commonly expressed in monetary terms (*expected monetary value*) as money is often the overriding decision-making criterion (Pollard et al 2004). Utility functions can be used to convert the attributes into a common scale, which are then aggregated into a final score. Once the utility function has been developed decision alternatives are ranked from the one with the highest utility to the lowest, and the alternative with the highest utility is considered to be the alternative that best maximises the decision maker's utility (*expected value*) and therefore the best desirable outcome. For decisions involving more than one decision maker, an aggregate of the decision makers' utility is used. Different stakeholders in the group decision-making may have different utility weights, depending on the value of their preference to the decision outcome (Kangas 1992).

This method of deriving utility functions, known as the Simple Weighted Average (SWA), is an example of an *additive* utility function because it involves adding the weighed utilities of individual attributes. An additive utility function assumes the independence of preferences form each other, and may not truly reflect the nature of individual preferences (Levin and McEwan 2000). This is a compensatory method, as the independence of preference allows weaker performing criteria to be compensated for by stronger performing criteria. Many other methods exist for deriving utility functions, including: Geometric Averaging (GA), Ordered Weighted Average (OWA) and also non-additive methods like the Choquet integral (Giove et al 2009).

MAUT is complex in practical application, and a simpler variant, the Multi-Attribute Value Theory (MAVT) is more commonly used. MAVT recognises the importance of criteria weights in decision making (Dyer and Sarin 1971, Cho et al 2003). However both the MAUT/VT methods are very complex, and due to their practical implementability (Linkov et al 2005) have been simplified into other derivatives such as SMART, SMARTER, SWINGS and TOPSIS (Edwards 1977, Winterfeldt and Edwards 1986, Hwang and Yoon 1981). These variants have been demonstrated to be not only more robust, but also replicate decisions made from the more complex MAUT/VT analyses with a high degree of confidence (Linkov et al 2005). In some cases these variants have even been shown to be more accurate than both MAUT/VT as they are not only easier to understand, but have more realistic scores and trade-offs.

Most MCDA applications to environmental management decision problems are based on the MAUT and its variants, and have been widely used for contaminated land management

(Linkov et al 2004). Examples of these applications include – site specific ecological risk assessment for contaminated lands (Critto et al 2007), site prioritisation in sediment management (Alvarez-Guerra et al 2009), for the development of Decision Support Systems (DSS) and Decision Support Tools (DST) (Monte et al 2006, Sorvari and Seppälä 2010, Critto et al 2002 and Sullivan et al 2008). The majority of contaminated land management applications are within the broad areas of managing environmental impacts and stakeholder involvement however, and very little effort has been made to apply MCDA to risk analysis or integrated management (Kiker et al 2005).

3.5.2 Outranking method

Unlike MAUT methods, outranking methods (Roy1973) do not assume that a single best alternative can be identified. The main objective of outranking is to provide decision makers' with a simple method using realistic preference modeling for the selection of good alternatives and to rank the alternatives (Parsaei et al 1993). Outranking does not require all criteria to be in a single unit and allows options to be classified as incomparable. Another important advantage of the outranking method, especially with respect to the utilitarian approach is its ability to deal with ordinal and more or less descriptive information on the alternative plans to be evaluated (Kangas et al 2001).

There are two main methods for solving outranking problems – the ELECTRE[21] and the PROMETHEE[22] methods. ELECTRE is suitable for decision problems involving at least three and at most 13 criteria (Roy 1991). The ELECTRE method consists of building a

[21] ELimination Et Choix Traduisant la REalité (*English*: *Elimination and choice expressing reality*)
[22] Preference Ranking Organization Method for Enrichment Evaluations

relation S_j (*the outranking relation*) from a finite set of alternatives A, and a set of criteria C for assessing the alternatives. Each criterion j is assigned some weight W according to its relative importance, where the sum of weights of all criteria equals 1. The outranking relation S_j is associated with each criterion j over the set of alternatives A.

The outranking relation is used to compare the performance of two alternatives at a time to identify the extent to which a preference for one over the other can be asserted (Linkov et al 2006b). For example, given two alternatives A and B, the relation $S(A,B)$ denotes alternative A is at least as good as alternative B with respect to all criteria. The relation $S_J(A,B)$ denote alternative A is at least as good as alternatives B with respect to criterion *j*. Dominance occurs when one alternative performs better than another on at least one criterion and no worse than the other on all criteria. Other types of relationships that exist between alternatives:

- Alternative A is preferable to alternative B if $S(A,B)$ is not the case that $S(B,A)$
- Alternatives A *and* B are indifferent if $S(A,B)$ *and* $S(B,A)$
- Alternatives A *and* B are incomparable if neither $S(A,B)$ *or* $S(B,A)$

Outranking relationships are evaluated using concordance (*Eq. 3*) and discordance (*Eq. 4*) indices (Raj and Kumar 1986). The concordance index $C(i,j)$ is the weighed measure of criteria i and j for which criterion i is preferred to criterion j, or for which i and j are equally preferred. $C(i,j)$ can therefore be considered as a measure of the decision makers preference of i over j. The concordance index is defined as:

$$C(i, j) = (W^{+} + 0.5W^{+}) / (W^{+} + W^{==} + W^{-}) \qquad (\textit{Equation 3})$$

where W^{+} is the sum of the weights for which i is preferred over j; $W^{==}$ is the sum of weights for which i and j are equally preferred, *and* W^{-} is the sum of weights for which j is preferred over i. The discordance index $D(i, j)$ is a measure of the dissatisfaction of choosing i over j. An interval scale common to all criteria is used for comparing the dissatisfaction caused by different levels of criteria. Each criterion can have a different range of scales, for example ordinal scale for qualitative criteria (*best to worse*). A normalised discord interval is calculated for each criterion where alternative j is preferred over i (Raj and Kumar 1986). The largest value of the normalised discord interval is the discordance index, which is defined as:

$$D(i, j) = (\textit{max. interval where } i < j) \,/\, (\textit{total range of scale}) \qquad (\textit{Equation 4})$$

The other commonly used outranking technique PROMETHEE is based on the ELECTRE method, and is comparably simpler, clearer and more stable (Parsaei et al 1993). The PROMETHEE method begins by defining an aggregated *preference index* and *an outranking flow*. The preference structure of PROMETHEE is based on pairwise comparisons of alternatives. A positive outranking flow implies an alternative is *outranking* all the others, and a negative outranking flow implies an alternative is *outranked* by all the others (Brans 1984). The alternatives that are not dominated by any other are called *efficient solutions* (Brans and Mareschal 1983). Dominance occurs in PROMETHEE the same way it does with ELECTRE. Environmental management is considered the most popular application of the PROMETHEE method**s (**Behzadian et al 2009).

Both the ELECTRE and the PROMETHEE methods have been widely used to support contaminated land management decision-making. Examples of applications of ELECTRE includes the remediation of petroleum contaminated land (Balasubramaniam et al 2007), the environmental impact assessment of a contaminated power plant (Cloquell-Ballester et al 2006), and for real choice process of a solid waste management system (Hokkanen et al 1995). Examples of the application of PROMETHEE include management of contaminated sediments (Linkov et al 2006a), decision support of watersheds (Hermans and Erickson 2007), and for incorporating stakeholder values in the management of contaminated sediments (Rogers et al 2005).

3.5.3 The analytical hierarchy process

The Analytical Hierarchy Process (AHP) is another MCDA method for decision-making under conditions of uncertainty (Saaty 1977). The AHP is based on four clearly defined axioms – (*i*) transitivity; (*ii*) reciprocity; (*iii*) dependence of a lower level element on the adjacent higher level element; *and (iv)* homogeneity that is characteristic of people's ability for making comparisons among things that are not too dissimilar with respect to a common property, and therefore the need to arrange them within an order preserving hierarchy (Saaty 1986).The AHP uses *objective functions* to aggregate the different facets of a decision problem with the main goal of selecting the alternative with the greatest value of the objective function – *i.e*. the alternative that maximises the objective function (Linkov et al. 2006*a*). Unlike MAUT, the AHP neither assumes transitivity (or the stronger condition of consistency) nor does it include strong assumptions of the usual notions of rationality

(Saaty 1986). The main use of the AHP is the resolution of choice problems in decisions involving multiple criteria, using three principles for problem solving: (*i*) decomposition; (*ii*) comparative judgments; *and* (*iii*) synthesis of priorities. Decomposition is used to systematically structure the decision problem into a top-down hierarchy of goals, criteria and alternatives to capture the basic elements of the decision problem, with the goal at the top followed by criteria and alternative (*Fig 3.4*). Elements in each level of the hierarchy must be homogeneous, decreasing in size from the top to the bottom level of the hierarchy.

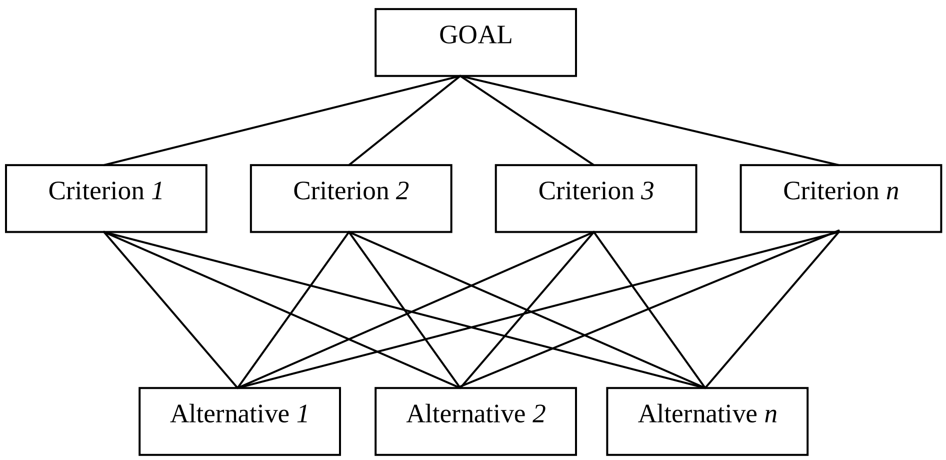

Figure 3.4 – Schematic of the Analytical Hierarchy Process

Both quantitative and quantitative criteria can be used to derive ratio scales for decision elements that share a common parent element at each hierarchical level. Decision makers' use comparative judgments to judge the relative importance of lower level elements with respect to the overall objective of a higher level using pairwise comparisons by asking questions like "with respect to criterion ***x***, how much more important or dominant is alternative *a* to *b*?". In cases where no measurement scale exist, pairwise comparisons are judged using the Saaty scale of absolute numbers which is used for assigning numerical values to both quantitative and qualitative judgements (*Table 3.4*).

Table 3.4 – The Saaty fundamental 9-point scale for comparative judgements

Intensity of importance	Definition	Explanation
1	Equal importance	Contribute equally to the objective
3	Moderate importance	Slightly favour one objective over another
5	Strong importance	Strongly favour one objective over another
7	Very strong importance	Favoured very strongly over another; dominance demonstrated in practice
9	Extreme importance	Evidence favouring one objective over another is of the highest possible order of affirmation
2, 4, 6, 8	For compromise between above values	Sometimes one need to interpolate compromise judgment numerically

* *When only two objects are compared it may be desirable to expand the interval 1, 2 (from equal to slight importance) by inserting the values, 1.1, 1.2, . . ., 1.9, starting with 1.1 as very slight, 1.2 as slight, 1.3 as moderate, etc.*

The pairwise comparisons are recorded in a comparative matrix A *(Eq.5)*, which must be both positive and reciprocal. The comparison matrix is said to be fully consistent if it is transitive: $a_{ij} = a_{ik}.a_{kj}$ where i, *j*, and *k* are alternatives in the matrix and reciprocal: $a_{ij} = \frac{1}{a_{ji}}$. The comparative matrix is used to derive ratio scales by computing the priorities of all elements at each hierarchical level based on their relative importance to every other element in their hierarchy, with respect to a common parent element, *i.e*. criteria are compared with respect to goal, sub-criteria to each of their parent criteria, and alternatives with respect to each sub-criterion, by normalising each column of the matrix to derive the normalised the priority vector. Many different methods are used for deriving priorities from comparison matrices, which generally fall into two groups – the Eigen value methods and the Geometric mean methods (Ishizaka 2004). Each method has its benefit and limitations, depending on the size of the matrices and the decision problem itself (Saaty 1990).

$$A = \begin{pmatrix} 1 & \dots & a_{ij} & a_{1n} \\ \dots & 1 & \dots & \dots \\ \dfrac{1}{a_{ij}} & \dots & 1 & \dots \\ \dfrac{1}{a_{1a}} & \dots & \dots & 1 \end{pmatrix} \qquad (\textit{Equation 5})$$

The AHP provides a method of calculating decision makers' inconsistency, which may arise from comparative judgements. A Consistency Index (*CI*) is used to determine whether decisions violate the transitivity rule, and by how much. Knowledge of inconsistency enables one to determine those judgments which need reassessment (Saaty 1986). A threshold inconsistency value of 0.10 is deemed acceptable, but if it is more than that then the consistency of the matrix can be calculated. The AHP can be used for group decision-making, and the judgments of each individual are aggregated to derive collective judgements, which must satisfy the reciprocal property.

The AHP method has become a major MCDA technique with a wide range of multidisciplinary applications, including application to many high level Government and corporate decision problems (*e.g.* Saaty and Vargas 2000, Chou et al 2007; Srdjevic 2007;

Wong and Li 2008). Its application to contaminated land decision-making has been limited however, and most of reported AHP applications are for conservation and natural resource management (*e.g*. Kangas 1994; Wolfslehner et al. 2005), site selection (Hill et al 2005, Moeinaddini et al 2010), land use planning (Ananda and Herath 2008; Dai et al 2001), water resources management (*e.g.* Jaber and Mohsen 2001; Thapa and Murayama 2008; Willett and Sharda 1991) and catchment management (Hill et al 2005).

3.6 CONCLUSION

Different decision-making techniques are used for structuring and facilitating contaminated land management decision problems. Due to the complexity of contaminated land management, decision-making techniques used in isolation often over simplify the decision situation and misrepresent it. Decision-making techniques also offer little guidance on how to integrate or judge the relative importance of information from each source (Kiker et al 2005), and do not provide structured means of arriving at optimal decision outcomes (Linkov et al 2006a). Furthermore using only one decision criterion cannot be regarded as a sufficient management approach (Janikowski et al 1998). It is widely acknowledged that economic, environmental and social criteria should be considered at the minimum for sustainable management of contaminated land.

Formalised and structured methods like MCDA provide a means of evaluating these multiple criteria. MCDA methods have been widely used to support contaminated land decision-making, and have been shown to offer significant improvements in the decision-making process (Bridges et al 2006, Linkov et al 2006b). MCDA methods do not all yield the same outcome for the same decision problem however, and some methods can yield the best alternative of one method as the worst of another and therefore give different results for the same decision problem (Cho et al 2003, Zanakis et al 1998). This inconsistency between the MCDA methods results from differences in the preference estimation process and in the calculation technique used (Kangas et al 2001).

Is there a better MCDA method for contaminated land decision-making? Several works comparing MCDA methods conclude there is no better method or worse method, and the successful application of any method depends on the decision problem and how it is formulated (*e.g.* Cho et al 2003, Malczewski 1999, Zanakis et al 1998, Olson et al 1999 *and* Belton 1986). The effective use of any method depends on the decision situation and how the decision problem is formulated. MAUT and AHP are suitable for situations where a single outcome is needed. MAUT and the AHP have very similar axioms, but differ on the methods of eliciting judgements on decision maker(s) preferences and in methods used for weighing criteria. The AHP also does not strictly adhere to conditions of transitivity and provides a means of judging inconsistency in judgements. Outranking methods are used for assessing the degree of dominance of one alternative over another, and are suitable in decision situations where no 'best' outcome exists because tradeoffs are too complex to judge relative importance between them.

4 DECISION SUPPORT SYSTEMS FOR CONTAMINATED LAND MANAGEMENT

4.1 INTRODUCTION

The modelling of environmental decision problems using Multi Criteria Decision Analysis (MCDA) methods is very common, both as a method for researchers and scientists to test theories to better understand the way systems function, and as a predictive or forecasting tool for better and quicker assessment of complex environmental systems (Rizzoli and Young 1997). MCDA solutions only model one site at a time however, and the solutions often involve a steep learning curve. Decision Support Systems (DSS) are used to encapsulate the MCDA decision models by codifying scientific and technical knowledge; expert judgement and policy requirements into stored process with the aim of providing concise representation of the decision problem (Bardos et al 2001).

In recent years DSSs have been successfully developed for contaminated land management to support consistent, rational, and transparent decision-making that is reproducible and therefore justifiable (Bardos et al 2001). These have varied from straightforward information systems about pros and cons of remedial options to formalised weighting systems that have been presented in different formats from simple diagrams derived from standards and regulations, to software based systems, developed as support or expert systems with varying degrees of success in practical use (CLARINET 2002b). Contaminated land DSSs are a subset of environmental DSS (EDSS), which are DSS containing at least one component supporting human decision-making about an environmental issue (Swayne et al 2000). Environmental problems are characteristically complex, dynamic, spatially distributed, multidisciplinary and highly non-linear; and usually controversial in the socio-economic domain. This is because many of the governing processes are not directly observable and therefore not easily understood (Fedra 2000). Three levels of complexity are associated with environmental management problems (Sànchez-Marrè et al 2008): (*i*) simple, low uncertainty problems for which models can provide accurate system descriptions; (*ii*) problems with higher degrees of uncertainty, models cannot provide satisfactory descriptions; *and* (*iii*) truly complex problems where much uncertainty exists, and where issues at stake reflect conflicting goals.

Although advanced mathematical and numerical models have long been used to support environmental management decision-making, not all decision support capabilities can be delivered using models alone, and a range of other tools for data analysis and presentation are required (Rizzoli and Young 1997). This makes necessary the development and application of new tools capable of processing not only the numerical aspects (models), but also experience from experts and wide public participation, which are all needed in decision-making processes (Poch et al 2003). It is also widely accepted that taking decisions in isolation is no longer sufficient, and there is a need to integrate models and tools into systems (Pollard et al 2001). DSS are very promising because of their ability to integrate different frameworks, architectures, tools and techniques to solve high level complexity (Poch et al 2003). DSSs have many capabilities that make them suitable for supporting these types of decision situations, including:

- Solving problems with varying degrees of structure (*structured, semi structured and* *unstructured decision problems*).

- Facilitating different types of decision-making, such as data analysis and retrieval, forecasting, planning, resource allocation, choice *etc*.
- Supporting both individual and group decision-making
- Supporting different stages/phases of the decision-making process.
- Supporting collaboration between decision makers.
- Handling large amounts of data.
- Handling both quantitative and qualitative information.
- Providing access to different data sources, formats, and types.
- Integrating different frameworks, technologies and tools for complex data analysis.
- Supporting with different problem solving methods, including optimisation, satisficing and heuristic methods.
- Providing insights into decision problem and opportunity for learning and training.
- Presenting of results in different formats, such as tables, graphs, reports *etc*.
- Providing transparency and justification of the decision-making process, thereby increasing confidence in decision outcomes.
- Supporting decision maker(s) with improved, consistent and timely decisions.
- Overcoming cognitive limitations of problem solving, data processing and storage.
- Once developed, can be repeatedly used for the same type of decision problem.

4.2 DECISION SUPPORT SYSTEMS

Although the history of the implementation of DSS begins in the mid 1960s, there is still no clear consensus on its definition or what it should do, and different researchers define it from different vantage points (Power 2007). Gorry and Scott-Morton (1971) coined the term DSS by arguing that Management Information Systems (MIS) primarily focus on structured decisions and proposed that information systems that support semi-structured and unstructured decisions should be termed Decision Support Systems (Power 2007). They went on to define DSS as "*interactive computer-based systems that help decision makers utilise data and models to solve unstructured problems*". Keen and Scott Morton (1978) defined DSS is a "*computer-based support system for management decision-makers dealing with semi structured problems*". Sprague and Carlson (1982) defined DSS as a "*class of information system that draws on transaction processing systems and interacts with the other parts of the overall information system to support the decision-making activities of managers and other knowledge workers in organisations*". More recently Power (2007) defined DSS as "*a type of interactive computer-based information system that supports decision-making activities and helps decision makers identify and solve problems, using different types of technologies, data, knowledge and/or models*".

Drawing on the various definitions above (summarised in *Table 4.1*), a DSS can be defined as interactive computer-based systems that: (*i*) utilise data and models to support rather than replace decision makers, *and* (*ii*) have decision-making capabilities as a component of problem solving, *and* (*iii*) facilitate the decision-making process, thereby focusing on the *effectiveness* rather than *efficiency* of decision outcomes (Eom 2001). DSS are not decision-making tools as they only assist with the decision-making process by providing knowledge to support decision-making. There are many variations of DSS, ranging from spreadsheet applications to large-scale computer-based modelling, from Executive Information

Systems[23] (EIS) to facilities for individual or group decision-making, and to hypertext storage and search systems to intelligent mechanisms such as Expert Systems[24] (ES). DSS differ in several important ways from their forerunners, data processing systems and Management Information Systems[25] (MIS), which were designed for processing large amounts of data and record keeping (Holsapple et al 2000).

Table 4.1 – The different contexts of DSS definition (after Turban and Aronson 2001)

SOURCE	DEFINITION CONTEXTS
Little (1970)	System function, interface characteristics
Gorry and Scott Morton (1971)	Problem type, system function (*support*)
Alter (1980)	Usage pattern, system objectives
Moore and Chang (1980)	Usage pattern, system capabilities
Keen (1980)	Development process
Sprague and Carlson (1982)	System functions (*type of support*)
Bonczek et al (1996)	System components (*degree of procedure*)
Power (2007)	System components (*dominant component*)

DSS have traditionally had two main tasks: (*i*) selecting information from available data sets and making it available for analysis; *and* (*ii*) building simple analytical models and applying them to data to examine the consequences of the model (Swayne et al 2000). DSS are best suited to supporting restricted but well understood application areas with well understood decision-making processes such as contaminated land management. DSS have a

[23] Executive Information Systems are designed to facilitate and support executive decision-making.
[24] Expert Systems are intelligent systems that encapsulate domain knowledge and expert judgement.
[25] Management Information Systems are designed to facilitate and support structured decision-making.

wide range of application area and have been successfully developed for different problems, including: medical diagnosis, financial management, business and organisational support, agriculture, forestry, natural resource management, water resources management, sustainable development and environmental decision-making (Kersten et al 2007).

4.2.1 *Taxonomy of decision support systems*

Due to the lack of consensus with definition, there is no definitive classification of DSS (Turban and Aronson 1995). Different classifications have been proposed over the years, including Anthony (1965), Simon (1977), Alter (1980), Bonczek et al (1980), Hackathorn and Keen (1981)**,** Sprague and Carlson (1982), Hättenschwiler (1999), and Power (2007).

Table 4.2 – The Gorry and Scott-Morton (1971) DSS classification with Simon (1977) classification in the left column and Anthony (1965) in the top row.

	OPERATIONAL CONTROL	MANAGERIAL CONTROL	STRATEGIC PLANNING	TECHNOLOGY SUPPORT
STRUCTURED DECISIONS	*e.g.* Accounts receivable; Order entry	*e.g.* Budget analysis; Short-term forecasting; Personnel reports	*e.g.* Investment analysis; Venue location	*e.g.* MIS; Models; Transaction processing
SEMI-STRUCTURED DECISIONS	*e.g.* Production scheduling; Inventory control	*e.g.* Credit evaluation; Project Scheduling	*e.g.* Mergers and acquisitions; New product planning	*e.g.* DSS
UNSTRUCTURED DECISIONS	*e.g.* Selecting or buying products; Approving loans	*e.g.* Negotiation; Executive recruiting	*e.g.* R&D planning; New technology development	*e.g.* DSS; EIS; Machine learning
TECHNOLOGY SUPPORT	*e.g.* MIS; Management Science	*e.g.* Management Science; DSS; ES; EIS	*e.g.* ES; EIS; Machine learning	

Anthony (1965) broadly classifies DSS according to the type of support they provide, which could be managerial, strategic or operational. Simon (1977) classifies DSS according to the degree of structure of the decision problem they support: highly structured, semi-structured and highly unstructured. Gorry and Scott-Morton (1971) proposed a classification based on both the Anthony (1965) and Simon (1977) classifications to include technology support (*Table 4.2*). The most commonly used classification is Alter's, which classifies DSS according to the degree of problem solving complexity and generic operations they can perform, into seven distinct groups. These are (*from lower to higher complexity*) (Alter 1979):

- *File drawer systems* that provide access to data items.
- *Data analysis systems* that provide *ad hoc* analysis of data files.
- *Analysis information systems* that provide *ad hoc* analysis involving multiple databases and small models.
- *Accounting and financial models* that provide standard calculations that estimate consequences of possible actions.
- *Representational models* for estimating the consequences of particular actions.
- *Optimisation models* for calculating optimal solutions to a combinatorial problem.
- *Suggestion models* for performing calculations that generate a suggested decision.

Bonczek et al (1980) classified DSS according to the degree of the procedurality of their data handling and modelling capabilities. Hackathorn and Keen (1981) classified DSS according to the type of support they provide – personal, group or organisational support. Hättenschwiler (1999) classified DSS according to the level of support they provide into active, passive and cooperative. The Power classification (Power 2007) is the most comprehensive and up to date, and classifies generic DSS into five categories:

- *Model driven DSS* are systems that provide access to and manipulation of simulation or optimisation models, and usually support complex analysis or choice between different options. Many of the early DSS are reported to be model driven.
- *Data driven DSS* are systems that provide access to manipulation of internal and sometimes external company and real-time data, and are used for querying databases to seek specific answers for specific purposes. These include data warehousing, Executive Information Systems (EIS) and OnLine Analytical Processing[26] (OLAP), and are usually institutional system.
- *Communication driven DSS* are systems that use network and communications technologies to facilitate decision-relevant collaboration and communication. This also includes software that supports group decision-making, which generally include forums and message boards, audio and video conferencing, groupware[27] and increasingly Voice Over IP[28] (VOIP).
- *Document driven DSS (also text oriented DSS)*: systems that use computer storage and processing technologies that manage, retrieve and manipulate unstructured information in a variety of formats.
- *Knowledge driven DSS*: systems that provide specialised problem-solving expertise

[26] OnLine Analytical Processing is a type of database application that facilitates data mining or trends and relationships in data.
[27] Groupware is software that is used by group of people working on the same information.
[28] Voice Over IP is a type of networking technology for transmitting voice conversations over a network using Internet Protocol (IP).

from stored facts, rules and procedures in a knowledge-base. Knowledge driven DSS are also known as knowledge-based DSS and cover a broad range of systems for example, all the other categories put together could make up knowledge-driven systems that serve different purposes.

On a technical level, Power (2007) further categories DSS into two: *enterprise-wide DSS* and *desktop DSS*. *Enterprise-wide DSS* are linked to large data warehouses and serve many managers in a company. *Desktop DSS* are single-user small systems that reside on individual manager's PC (Power 2007). Other discipline specific classifications have been proposed in the literature, *e.g*. Rizzoli and Young (1997) classifies EDSS into two:

- *Problem specific* EDSS are tailored to specific environmental problems, but can be applied to a wide range of different locations with the same problem.
- *Situation and problem specific* EDSS are tailored to both specific problems and specific locations, and cannot be easily modified for use in other locations, even within the same problem domain.

4.2.2 Architecture of decision support systems

Different researchers have proposed frameworks and architectures for developing generic DSS, including: Sprague (1980), Bonczek et al (1981), Marakas (1999), Turban and Aronson (2001) *etc*. Sprague (1980) proposed the main components of a DSS should be: (*i*) a database; (*ii*) a model base; *and* (*iii*) an intermediate software system that interfaces with the DSS. Bonczek et al (1981) proposed the components should include: (*i*) a language system; (*ii*) a knowledge system; *and* (*iii*) a problem processing system. The language system is the linguistic facilities made available to the decision maker(s). The knowledge system is the knowledge about the problem domain, essentially a knowledge-base. The problem processing system is the interface between the language system and the knowledge system. Marakas (1999) proposed an architecture comprising five distinct components: (*i*) a data management system; (*ii*) a model management system; (*iii*) a knowledge engine; (*iv*) a User Interface (UI); *and* (*v*) the users. More recently Turban and Aronson (2001) proposed an architecture with four core components consisting of: (*i*) a data management component; (*ii*) a model management component; (*iii*) knowledge management component; *and* (*iv*) a UI component (*Fig 4.1*).

The data management component includes a database or data warehouse[29] containing the relevant data/information for the decision situation, and a Database Management System (DBMS) for database manipulation. The model management component contains any model(s) used by the DSS, and is managed by a Model Management System (MMS). Models could be: (*i*) strategic for high level managerial support; (*ii*) tactical for allocating and controlling organisational resources; (*iii*) operational for supporting day to day activities; *or* (*iv*) have other functionalities according to the discipline or decision situation. The knowledge management component includes a knowledge-base and an inference engine, and provides intelligence to supplement the operations of the other components. The knowledge-base encapsulates domain expertise, and is an essential component of intelligent DSS (*also expert system or knowledge-based systems*).

[29] A data warehouse is essentially a database that is designed to facilitate database querying and analysis.

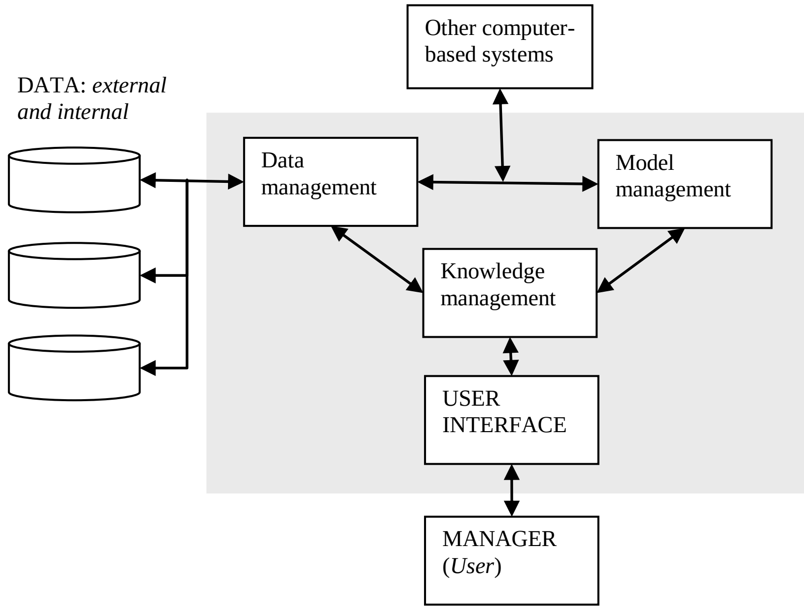

Figure 4.1 – Schematic of the components of a decision support system (Turban and Aronson 2001)

The use of Artificial Intelligence (AI) techniques to develop intelligent DSS provides direct access to distributed and multidisciplinary expertise. The flexibility of intelligent DSS makes them capable of supporting learning, complex analysis and decision support. Their integration with other DSS tools in a single system provides advanced decision support (Cortés et al 2000). Finally the UI component provides the means for decision maker(s) (the users) to interact with the DSS. For many decision makers' the UI is the system. In many ways the UI is the most important component, as it heavily influences how users perceive and use the DSS. Much of the design and development effort should therefore ideally focus on building the UI (Power and Sharda 2009).

Both experience and empirical evidence indicate that design and implementation issues vary for the different types of DSS, and different DSS may require specialised modelling or database components for example. Differences in the types of usage – *e.g.* individual or group decision-making may also create complex implementation issues (Power and Sharda 2009). Different disciplines will also require a different approach to decision support, and as such discipline specific architectures have also been proposed, for example Abel et al 1992, Frysinger 1995, Fedra 2000, Cortés et al 2000, Poch et al 2003, Denzer 2005 and Matthies et al 2007 for EDSS.

Environmental problems specifically require a different approach to decision support for two fundamental reasons: (*i*) complexity of the environmental systems; *and* (*ii*) the changing nature of the decision making process itself (Fedra 2000). Swayne et al (2000) have identified three aspects of EDSS that make them significantly different from traditional DSS. These are: (*i*) the scale of the data sets; (*ii*) the complexity of the data sets; *and* (*iii*) the association of physical reality of the data sets. However even with the understanding of these EDSS requirements, EDSS generally have no fixed architecture (Swayne et al 2000) and could consist of four core components (Denzer et al 2005):

- a database component with sophisticated DBMS capabilities;
- a knowledge representation component;
- a component that deals with problem processing; *and*
- a simple, powerful and user friendly Graphical User Interface (GUI).

These components can be tailored to specific environmental problems and additional components for data and results visualisation and geospatial capabilities could be built-in according to needs. Denzer (2005) posits the key characteristics of EDSS are:

- The complexity of the environmental management system itself.

- Complex data management, with time and space-related data that is often incomplete, fuzzy or of the wrong scale needed for a given task.
- Most problems cannot be solved by a single tool or method, and therefore requires the integration of different tools, methods, techniques using different algorithmic and/or data management strategies into holistic solution for end users.
- Tools/software are developed using a different algorithms and programming paradigms resulting in complex systems from different domains of information technology and software development.

4.3 REVIEW OF DECISION SUPORT SYSTEMS FOR THE MANAGEMENT OF CONTAMINATED LAND

DSS have been successfully developed using a wide range of decision-making techniques and MCDA methods for a wide range of environmental management problems. Many EDSS applications for contaminated land management are reported in the literature, and these applications have been found to significantly improve the consistency, quality and timeliness of decision outcomes (Cortés et al 2000, Poch et al 2003). However a lot of these are in fact different models integrated to better visualise data or describe systems, and do not specifically address decision problems or help decision makers with making inevitable tradeoffs between criteria (Giove et al 2009). It is therefore necessary to highlight the difference between decision support information, tools and systems for use in decision-making. Bardos et al (2001) proposed a conceptual framework for information use in contaminated land decision-making emphasising ‘system’ as a totality of the decision processes (*Fig 4.2*). In the framework models are not considered DSS, but rather input;

tools and techniques represent component parts of the decision-making process; and '*system*' supports the totality of a particular decision-making process.

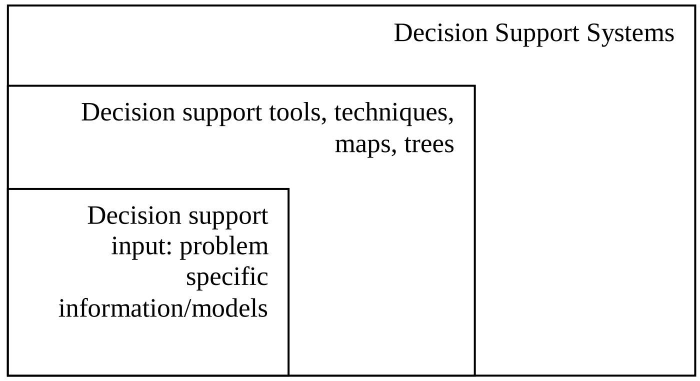

Figure 4.2 – Relationship between decision support information, tools and systems (Bardos et al 2001)

The DSS applications to contaminated land management can be broadly categorised according to their functionality as: group DSS for decision-making involving a team of decision makers', spatial DSS for decision-making involving spatial analysis, intelligent or knowledge-based DSS for analysis of complex decision problems and Web-based DSS which are a new DSS paradigm that makes possible the implementation of group, spatial, intelligent and/or knowledge-based DSS application deployed over the Web.

4.3.1 Group decision support systems

A group DSS (*also collaborative DSS or groupware*) is a collection of software, hardware, and procedures designed for the automated support of decision-making in a group environment, which could include brainstorming or assigning preferences to decision alternatives. As a high proportion of managerial decision-making is undertaken in a group environment, group DSS have been found to be beneficial as they can decrease the amount of time necessary for meetings by over fifty percent. This is sufficient incentive for many organisations for investing in group DSS (Aiken et al 1995). Group DSS can foster collaboration, communication, and negotiation amongst group participants, and hence arriving at effective decisions quicker. Group DSS are best suited to complex and ill structured decision problems involving a large group of geographically distributed decision-makers as small groups rarely justify the investment.

Group DSSs are distinguished from individual DSS in terms of their functional purpose or components, with the fundamental distinction being supporting a group of decision makers as opposed to an individual decision maker (DeSanctis and Gallupe 1985). Another distinction is with system components, with group DSS having the same components as that of a DSS, but with specific communication component for collaboration and communication between decision makers. Group DSS have many advantages over other DSS or non-automated group support, including: providing anonymity of input, conflict resolution, fostering collaboration between decision makers', reinforcing positive group behaviour, providing incentive in participation in group decision making activities, automated record keeping and reducing redundancy in decision-making in multidisciplinary environments (Aiken et al 1995). The participants involved in the group decision-making can also be in different environmental setting depending on the location or number of participants, which could be: a decision room for face to face meeting with a small group of decision makers in the same room, a network involving a small group of decision makers dispersed over a Local Area Network[30] (LAN), a legislative session involving a large group of decision makers in a face to face meeting, and/or computer conference involving a group of geographically dispersed decision-makers (Aiken et al 1995).

Different techniques are used for supporting group decision-making, including: brainstorming, focus groups, the Nominal Group Technique (NGT), and the Delphi method. Brainstorming and focus groups use qualitative research methods such as NGT and Delphi method to derive individual decision maker's preferences, which are then aggregated to support group decision-making. NGT allows decision-makers to individually address the decision task. The individual outputs are then tallied and duplicate outcomes are eliminated. The remaining solutions are ranked by decision makers individually and the final score of each decision is summed and the decisions ranked from the highest scoring to the lowest. The Delphi method is used for obtaining group consensus from individual decision makers' preferences through series of individual questionnaires. The results of the questionnaires are used to prepare the next questionnaire containing information and opinion of the group, and the decision makers are encouraged to reconsider or revise their decision in response to the information provided. The process continues until group consensus is achieved.

MCDA methods are used for group decision making where decision makers' agree on an explicit set of outcomes, but have different individual preferences (*utilities*) regarding the priority or importance of each outcome (Iz and Gardner 1993). The individual decision maker's preferences are aggregated using qualitative decision-making techniques like NGT

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[30] A *Local Area Network* (LAN) is a communications network that connects devices (computers~) in close proximity, such as within an office building.

and Delphi method, or axiomatic preference aggregation methods such as the additive or multiplicative group utility functions (Keeney and Raifa 1976). Group DSS have been found to be especially beneficial for integrating information and resolving misunderstanding arising from geographically dispersed expertise, discipline specific techniques and processes, and/or linguistic expressions of decision-makers preferences and opinions (Lu et al 2005). Many Group DSS have been developed for contaminated land management, including SUDSS for land use planning aimed at accommodating public participation in all decision-making processes (Jankwowski and Stakis 1997), a multi criteria group DSS Web-HIPRE for participatory environmental modelling (Mustajoki et al 2004), and a group DSS RODOS for emergency management of nuclear or radiological accidents (Geldermann et al 2005).

4.3.2 Spatial decision support systems

Spatial DSS is another DSS paradigm that has been widely applied to contaminated land management decision problems. Spatial DSS integrate the two distinct disciplines – DSS and Geographical Information Systems (GIS) for advanced solutions for spatial decision problems, by providing a decision-making environment for complex and unstructured spatial decision problems (Sugumaran and Sugumaran 2005). Spatial decision problems characteristically involve a set of spatial alternatives with spatially variable consequences, involving multiple criteria that could be both qualitative and quantitative (Ascough II et al 2002). Spatial decision-making requires information on spatially distributed alternatives and the decision maker(s) preferences. The spatial decision-making process could also involve either the individual or a group of decision makers.

Standalone GIS is used for different purposes in contaminated land, including site characterisation, developing conceptual model for spatial analysis of pollutant-linkages or environmental impacts, risk assessment *etc*. GIS use: (*i*) maps to conceptualise and visualise the decision problem; (*ii*) overlays to define relationships; *or* (*iii*) modelling pollutant fate and transport. GIS can provide some decision support capabilities, albeit to a limited extent as they lack the capability for complex analysis of unstructured spatial problems (Segrera et al 2003). GIS also do not provide a means of assessing and choosing from competing alternatives and as such cannot to be used to support decision-making (Yan et al 1999). On the other hand, standalone DSS lack the capability of spatial analysis. Spatial DSS therefore extends the capabilities of both GIS and DSS to support both spatial analysis and decision-making process of complex, ill-defined, spatial decision problems. Spatial DSS provide a rational and objective approach to spatial decision analysis by supporting decision maker(s) with assessing and evaluating the consequences of the inevitable tradeoffs between decision alternatives (Wang and Cheng 2006).

Spatial DSS comprises two core components: (*i*) a GIS component for spatial analysis, *and* (*ii*) a decision support component for evaluating a decision maker's preferences (Ascough II et al 2002). The core components of a spatial DSS are: (*i*) a database component for storing information and a DBMS for manipulating the database; (*ii*) a model component that could include decision rules; *and* (*iii*) a UI component for decision makers to interact with the database, model and the rest of the system. Examples of spatial DSS application to contaminated land decision problems include an analytic tool for management of environmental pollution, and protection of natural resources (Diah 1997), a spatial DSS for the management of decommissioned contaminated nuclear sites in the UK (Hitchins et al

2005), HYDRA - a prototype grid-enabled spatial DSS for providing access to data for spatial analyses for improved decision-making in urban brownfield redevelopment (Birkin et al 2006), GISSIM - a system for management of petroleum contaminated land (Chen and Chakma 2002), SmartPlaces - a prototype spatial and expert DSS by for urban siting decisions to prevent redevelopment on potentially contaminated land (Thomas 2002), *SuSAP* - for risk assessment (Chaiudani et al 2007) *etc*.

4.3.3 Intelligent decision support systems

Intelligent DSS are a type of model-based DSS that use Artificial Intelligence (AI) techniques for solving complex decision problems. AI is the concept of applying human intelligence to machines, specifically computer systems. The use of AI and knowledge-based techniques in environmental modelling has increased with recognition of its potential (Chen et al 2008), and intelligent DSS such as knowledge-based systems (also sometimes referred as expert systems) have been developed to support with contaminated land management decision-making. AI is a multi-disciplinary field that applies theories from fields as diverse as mathematics, philosophy, psychology, neurology and biology to imitate key aspects of human intelligence such as learning, reasoning, problem solving, natural language, and knowledge representation and manipulation.

The most commonly used AI techniques for solving complex environmental decision problems are: Fuzzy Logic (FL), Artificial Neural Networks (ANN) and Genetic Algorithms (GA). These techniques simulate natural phenomena from different perspectives: ANN model biological nervous systems, GA uses a stochastic searching process based on mechanisms of natural selection and natural genetics, and FL simulates human decision making processes in a high level manner and is uses fuzzy set theory to model different types of uncertainties (Cheng and Ko 2006).

ANN consists of a network of interconnected processing elements (*neurons*) that work together in parallel for problem solving (*Fig 4.3*). ANN are able to derive meaning from complex or imprecise data, which can be used to extract complex patterns and detect complex trends. ANNs are able to learn, recall, and generalise from training patterns and are used for solving semi structured or unstructured problems, such as contaminated land management decision problems. They are capable of learning patterns of relationships in data from given inputs(s), generalising or abstracting results from imperfect data, and are insensitive to minor variations in input such as noise or incomplete data. ANN are useful in solving data intensive problems where the algorithm or rules required for problem solving are unknown or difficult to express (Chen et al 2008).

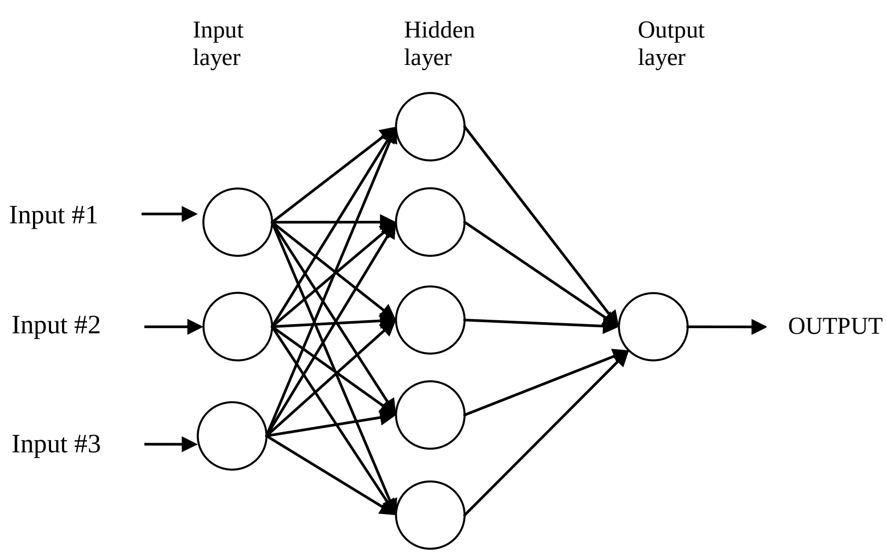

Figure 4.3 – Schematic of an artifitial neural network with three inputs, five hidden nodes, and one output

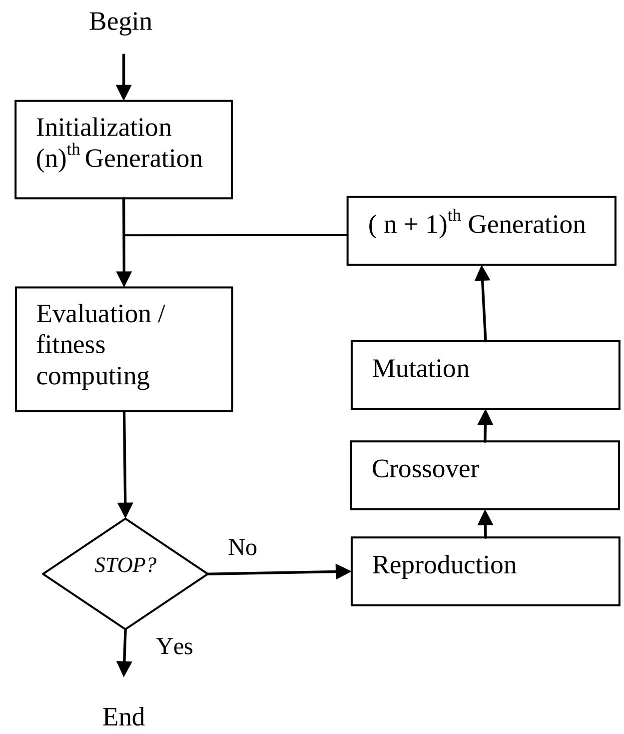

Figure 4.4 – Structure of a genetic algorithm

Like ANN, the GA concept is derived from nature, and is the computational equivalent of evolutionary phenomena. GA is an evolutionary algorithm used for solving optimisation or search problems by mimicking Darwinian natural selection and survival of the fittest. The goal of GA is to develop systems that can adapt by exposure to the environment, *i.e*. evolve with time. A GA consists of a population of individuals, each representing a possible solution to the problem. The evolution begins by randomly generating many individual solutions from the initial population. In each generation, the fitness of each individual in the population is evaluated, from which multiple individuals are stochastically selected based on their fitness and modified to form a new population. This is achieved by allocating each individual a fitness measure according the effectiveness of the solution it produces. The fittest individuals survive to the next generation, and the weak individuals are eliminated. The new population is used in the next iteration of the algorithm. The algorithm terminates when a satisfactory fitness level has been achieved, or the maximum number of generations have been produced even if a satisfactory solution has not been reached (*Fig 4.4*).

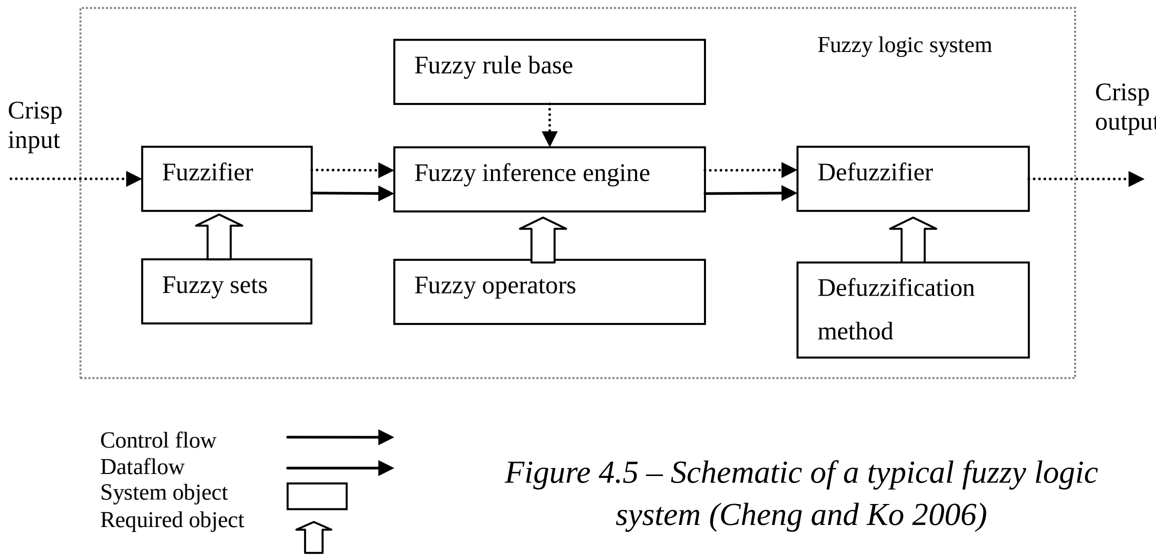

Figure 4.5 – Schematic of a typical fuzzy logic system (Cheng and Ko 2006)

FL is used for supporting high level decision-making processes using Fuzzy Set Theory (FST), as it allows the representation of vague and imprecise knowledge. FST is a generalisation of classical set theory where set memberships are graded, with the grade of membership in the set taking values in a unit interval or, more generally in a partially ordered set (Zadeh 1965). A Fuzzy Logic System (FLS) uses fuzzy membership functions and rules for reasoning using linguistic expressions, and contains four major components: a fuzzifier for converting input values, a fuzzy inference engine, a rule base, and an optional defuzzifier (Cheng and Ko 2006). A FLS works by first converting input values from crisp to fuzzy terms, which are then evaluated by the fuzzy inference engine using a set of predefined fuzzy rules from the rule base. These evaluations provide the basis of decision support. The fuzzy outputs are then aggregated into single fuzzy subset for each output variable. Finally an optional defuzzifier is used to translate output values into crisp value for end use (*Fig 4.5*). FLS provide a logical way of expressing natural language without losing semantic value, and have been found to be especially useful for assessing subjective information that is hard to quantify using traditional logic (Mendel 1995). FLS have been applied to contaminated land management problems – for example, to address uncertainties associated with site investigation (Heinrich 2000, Genske and Heinrich 2008), sampling and identification of hotspot (Özdamar et al 2000), modelling contaminant fate and transport (CLARA *c*2010), and for decision analysis (Mohamed and Côté 1999).

4.3.4 Expert systems and knowledge-based decision support systems

Although AI was primarily concerned with game playing, planning and problem solving in the early days, the most important application areas of AI are now centred on knowledge engineering, particularly expert systems (Giarratano and Riley 1989). Expert systems (ES) are a type of AI application that use reasoning techniques for decision-making. These systems encapsulate high level expertise that cannot easily be transferable, for problem solving and are essentially diagnosis machines (Pomerol 1997). The knowledge in Expert Systems (ES) may be either expertise or knowledge that is generally available from books, manuals or knowledgeable persons. The terms ES, Knowledge-Based Systems (KBS), Knowledge-Based Expert Systems (KBES) are often used synonymously, although there may not be expertise in the ES, only general knowledge. It is common practice to use ES to refer to KBS even when the knowledge in the system is not at the level of a human expert

(Giarratano and Riley 1989). The fundamental difference between KBS and ES therefore lies in the method of knowledge elicitation. ES are designed to be high performing, have adequate response times, and is reliable and understandable (Giarratano and Riley 1989).

Knowledge-based technology can be considered a term for the application of knowledge-based techniques, which may be used for developing either ES or KBS. This may be represented using different methods and techniques, and not only knowledge-based techniques. Knowledge-based techniques are suitable for problems that require heuristic or empirical solutions rather than mathematical solutions. Like FLS, ES and KBS use subjective information as opposed numerical data used by to conventional programming techniques (Toll 1996). In many ways, despite philosophical claims that systems based on manipulation of facts and inference rules cannot scale up to intelligent behaviour, ES and KBS are not only the most successful areas of AI (Crighton 2005), but are the fastest growing branches of AI in geotechnical and geo-environmental engineering (*Fig 4.6*).

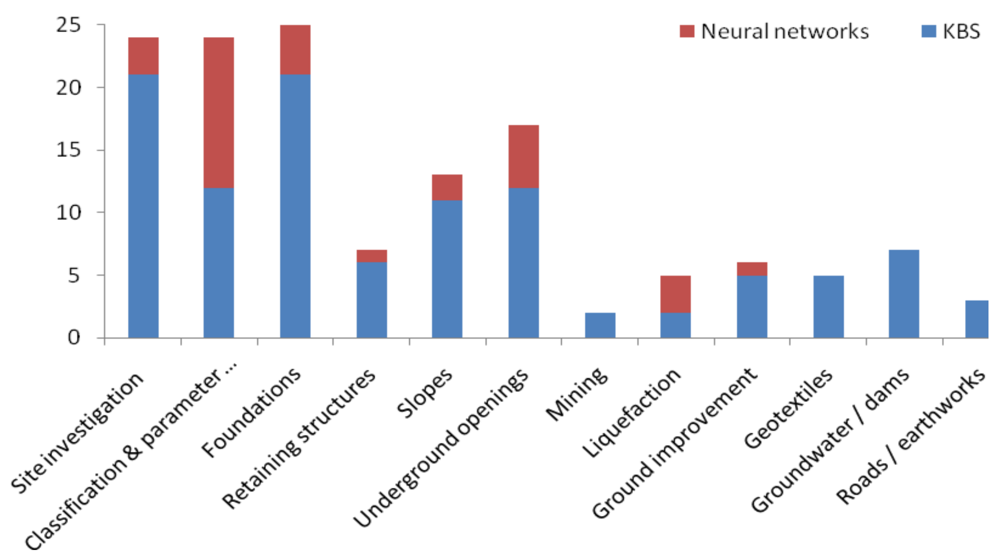

Figure 4.6 – Artificial intelligence applications in geotechnical engineering, classified according to the area of application (Toll 1996)

ES and KBS use knowledge and inference procedures to solve problems that are difficult enough to require significant human expertise for their solutions. These systems make extensive use of specialised knowledge to solve problems at the level of a human expert, who has knowledge or skills in a certain area (*the problem domain*) that are not known or available to most people, or can solve them much more efficiently. A problem domain is a specialised problem area such as medicine, engineering or finance. The expert's knowledge about problem solving is known as the knowledge domain, which is within the problem domain. The systems reasons or makes inferences in the same way that a human expert would infer the solution of a problem. ES are suited to problems with no algorithmic solutions and rely on inferences to achieve reasonable solutions. The development of ES or KBS begins with knowledge acquisition where the domain knowledge is gathered from experts and/or other sources and stored in a knowledge-base. This knowledge is heuristic in nature, and based on *rules of thumb* rather than absolute certainties (Cawsey 1994), and therefore an inference engine is used for processing the knowledge.

The knowledge of ES is represented in number of ways – it can be encapsulated in rules and objects. A common method of representing knowledge is in the form IF…THEN type-rules, such as, IF (*fact*) THEN (*match pattern and perform action*). If a fact exists, a pattern is matched and the action is performed. Newell and Simon (1972) demonstrated that much human problem solving or cognition can be expressed by IF…THEN type rules. Although this may seem simple, many significant real world expert systems have been built by representing knowledge or expertise in rules, such as the DEC XCON/R1 for configuring computer systems, DENDRAL for identifying chemical constituents from mass spectrograms, and MYCIN for medical diagnosis. KBS generally consist of two core components: (*i*) a domain-specific knowledge-base; *and* (*ii*) a domain independent inference engine (Vari and Vecsenyi 1988). The knowledge-base contains specialised domain specific problem-solving knowledge of one or more experts/disciplines (Power 2009). KBS encapsulates expert knowledge from sets of predefined facts, using either rule-based or case-based reasoning. Facts are a set of relations (*information*) that are known to be true, and are stored in the knowledge-base. The inference engine uses the information from the knowledge-base along with problem specific input data to generate useful information about a specific case.

In case-based reasoning, problems are matched to existing cases in the knowledge-base, and decisions are based on previous cases. Rule-based reasoning, on the other hand uses simple IF-THEN statements, with complex reasoning achieved by chaining rules together. Chaining is achieved by either forward or backward chaining methods. Forward chaining is a top-down approach that uses the information available and then infers rules until a desired goal is reached (*goal driven*), while backward chaining is the other way round that uses a bottom-up approach that starts with goals (*data driven*). Mixed chaining can also be used, combining forward and backward chaining.

An important feature of a KBS is that the inference engine is separate from the knowledge-base. This separation allows knowledge to be modified without having to change the computer code (Cawsey 1994). Uncertainty is frequently an issue within KBS, as simple rule-based systems may only use first order logic (a deterministic approach) in which outcomes are either '*True*' or '*False*', therefore not reflecting any uncertainty that may be present as a result of subjective knowledge that may not be precisely defined (Toll 1996).

4.3.5 Web-based decision support systems

The latest developments in DSS are Web-based (Power 2007), including a lot of recently developed DSS for environmental management, for example SMART*e* - a Web-based DSS for sustainable brownfield revitalisation (Vaga et al 2008) and OntoWEDSS - a Web-based DSS for wastewater management (Ceccaroni et al 2004). As the internet increasingly becomes the primary source of information, organisations and users are increasingly relying on the Web for decision-making processes (Jarupathirun and Zahedi 2007). In general all types of DSS can be implemented using Web technologies. The evolution and growth of the Internet, the globalisation of industries and the growing societal awareness of the effects of our activities on the environment have led to industries having to modify the way they operate by using technological innovations (Lago et al 2007). This has allowed innovative and powerful Web-based DSS to be developed, using analytical and visualisation tools mostly as model-based DSS (Black and Stockton 2007).

There are several advantages in developing Web-based DSS, including reducing technological barriers and making it easier and less costly to make decision relevant information and models available to decision makers' **(**Power et al 2000). Whilst traditional DSS require software installation on individual PCs and/or the use of proprietary software, Web-based DSS are available over the Internet, requiring no installation. All processes execute on a centralised remote server managed by the service provider, and all that is required is a thin-client (*e.g*. Web browser) for access. This paradigm of software delivery as a distributed *pay-as-you-go* service has obvious cost and practical advantages: organisations no longer have to commit to dedicated hardware and/or bespoke software.

Other benefits of using Web-based DSS over traditional DSS are information distribution and platform independence (Molenaar and Songer 2001). The ubiquitous nature of the Web and its simple and intuitive user interface has made possible the development of all types of DSS for contaminated land decision support on the Web such as spatial DSS (Carlon et al 2008, Goor et al 2003, Howard et al 2005, Monte et al 2009 *and* Sugumaran and Sugumaran 2005), AI based DSS (López et al 2008, Dixon 2005, Genske and Heinrich 2009, García et al 2006 and Chan and Huang 2003) and knowledge-based DSS (Avouris 1995, Martin and Toll 2006, Wilson 1987). Web-based GIS applications are particularly becoming widely available, making it easy to develop complex and integrated spatial DSS that represent and analyze spatial data using web browsers as a front end - for example, *WCDSS* - a prototype Web-based Spatial DSS for information exchange and distribution to help with collaborative decision making (Wang and Cheng 2006).

Many organisations offer free, powerful and interactive Web mapping services (*e.g.* Open Layers, Google™ Earth API, Google™ Maps API, NASA WorldWind SDK, and Microsoft® Bing Maps API) and visualisation suites like Google™ SketchUp (3D modelling) and Google™ Visualisation API (interactive charts). The Application Programming Interface (API) can be used to build complex DSS for contaminated land decision-making. Reputable organisations like British Oceanographic Data Centre, the UK Met Office, the US Alaska Volcano Observatory, and the British Antarctic Survey have all been using Google Earth to display and interpret live information (Simonite 2007), so they are quiet advanced for providing complex decision support. These developments are very promising for the development of Web-based DSS; however there are issues of data confidentiality and Web security that need to be addressed, especially with sensitive information processed and stored on third party servers at supposedly secure data centres that stakeholders might not necessarily want to share. Information on contaminated sites is highly sensitive and stakeholders might not want that publicly available, so subscribing to Web-based software is not always a viable option.

4.4 CONCLUSION

Although numerical and statistical simulations have long been used to garner insights into environmental problems, the complexity of such problems requires the application of new methods. EDSS are amongst the most promising solutions because of their ability to integrate different frameworks, architectures, tools and methods for solving higher level complexity (Poch et al 2003). Many MCDA methods have been used for solving complex contaminated land management decision problems, and many EDSS have been developed for contaminated land decision support with varying degrees of practical success (CLARINET 2002). Most of these DSS are experimental, developed by researchers to solve specific problems on *ad hoc* basis (Weistroffer et al 2006). Much interdisciplinary work remains to be done within the artificial intelligence, computer science (*GIS, statistical and mathematical modelling*) and environmental science communities (Sànchez-Marrè et al 2008) for better integration of systems and components.

MCDA methods enable considerations of multiple criteria such as social, economic and environmental factors for decision-making, especially in cases of uncertainty and data scarcity, such as in contaminated land management decision-making. A large majority of contaminated land DSS are AI-based. AI-based DSS often raise questions of the cognitive power and adequacy of intelligent systems in providing support (Radermacher 1994).

However philosophical arguments aside, AI-based systems have long been successfully applied in safety critical situations such as medical diagnosis and management of nuclear power plants for decades. Moreover the majority of the EDSS applications for contaminated land management use one or more of the AI techniques.

This chapter highlighted the wide range of opinions of what constitutes an EDSS, largely because EDSS are relatively recent and integrate multiple tools and architectures – especially within the context of contaminated land management. Although one may argue that a database management system could be used for decision support, today's consensus is that EDSSs must adopt a knowledge-based approach (Poch et al 2003). Many EDSS applications are reported in the literature, and a lot of them have been reviewed here. However a lot of them are in fact different models integrated to better visualise data or describe systems, and do not specifically address decision problems or help decision makers in making inevitable tradeoffs (Giove et al 2008). The majority of these DSS focus on risk assessment, technology selection and stakeholder involvement (Agostini and Vaga 2008), and rarely look at the whole contaminated land management process holistically. As all aspects of the management process are related and have a bearing on the final decision, there is a need to integrate the different models, software and tools into single portal for effective management. However, integration is a challenge as different components are developed using different methods, and may even be programmed using different programming languages. Despite these challenges however, DSS allows the relatively easy integration of disciplines from classical fields of all kinds of optimisation, to stochastics, decision theory, decision-making, decision support and so forth (Radermacher 1994).

5 FRAMEWORK FOR DEVELOPING DECISION SUPPORT SYSTEMS FOR CONTAMINATED LAND MANAGEMENT

5.1 INTRODUCTION

Many Decision Support Systems (DSS) have been developed to support with different levels of contaminated land management decision-making. However no well defined methodology or framework exists for the development of such systems (Sànchez-Marrè et al 2008). Although traditional approaches to software engineering[31] can be used, the development processes and the technical requirements of DSS differ in terms of development methods, tools, technical evolution and the type(s) of expertise required by the developers' and decision makers' (Gachet and Hättenschwiler 2003). The development process requires consideration of the decision problem to be solved, the appropriate decision-making method(s) required, the problem specific decision-making process, the decision-making level and the computational requirements (Black and Stockton 2009).

Developing a framework for development of DSS is helpful in structuring and identifying the relationships between all the constituent parts (Sprague 1980). Frameworks are also useful for providing consistency in decision-making for the same types of decision problems (Black and Stockton 2009). Several frameworks have been proposed for developing both generic and discipline specific DSS. One of the earliest and widely used generic framework was by Sprague (1980) which is based on: (*i*) the different levels of

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[31] Software Engineering is an engineering discipline concerned with all aspects of software development and delivery from specification, to design, development and deployment, in a timely and cost effective way.

technology that will be used over the development of DSS and the relationships between them; (*ii*) the developmental approach; *and* (*iii*) the different types of people involved with the development and use of DSS and the roles they each play. This and other early frameworks are useful for rapid development of problem-specific DSS on individual computers, but do not make provisions for integration with other tools or systems for decision support that are necessary for deploying contemporary DSS (Bui and Lee 1999). More recent frameworks such as Bui and Lee (1999) take these issues into account, and many other generic frameworks have been proposed, examples of which include Gachet and Hättenschwiler (2003), Mateou and Andreou (2008), *etc*.

5.2 THE NEED FOR A FRAMEWORK

The frameworks for developing environmental DSS (EDSS) include Cortés et al (2002), Poch et al (2003), and Sànchez-Marrè et al (2008). Cortés et al (2000) proposed an approach to developing EDSS involving five steps: (*i*) data gathering, interpretation and storage; (*ii*) diagnostic level involving modelling; (*iii*) decision support level using information from the database and models; (*iv*) presentation of decision support result to be used as decision-making knowledge; *and* (*v*) recommendation. Sànchez-Marrè et al (2008) proposed a cognitive-oriented approach to developing intelligent EDSS which involves three tasks: (*i*) analysis; (*ii*) synthesis; *and* (*iii*) prognosis. Analysis tasks involve data gathering, knowledge discovery and development of diagnostic models. Synthesis tasks involve aggregating alternate solutions from different diagnostics for solution generation. Prognosis tasks involve supporting decision-making from the aggregated solutions.

Despite these contributions however, no single framework dominates and the development of DSS is still distinctly an *ad hoc* process (Gachet and Hättenschwiler 2003). This is mainly due to the fact that the development of DSS is a multidisciplinary process involving knowledge of the DSS application area and techniques and tools from mathematics, Operations Research[32] (OR), computer science and software engineering. The different frameworks have also been developed from different perspectives, with emphasis on either decision support or software development processes. However this approach is inadequate because both perspectives are critical for DSS development and therefore need to be reconciled (Gachet and Hättenschwiler 2003).

Generic frameworks such as frameworks for developing EDSS, are helpful in structuring similar decision problems, for example environmental decision problems are similar in that they require consideration for the same economic, environmental and social criteria (Black and Stockton 2009). However generic frameworks still need to be adapted to specific application areas because of differences in decision problems, the type of information and knowledge that can be acquired, and the decision-making process (Poch et al 2003). As a result of this, a framework for developing contaminated land management DSS is required. This is due to the fact that the development of contaminated land DSS requires consideration of several specific factors, including: (*i*) comprehensive knowledge and understanding of the different decision-making methods and techniques that can be used; (*ii*) the contaminated land management decision-making process and its constraints; (*iii*) comprehensive understanding of the underlying multidisciplinary science governing contaminant behaviour and transport and the risks this poses to different types of receptors;

[32] Operations Research is a discipline that uses analytical methods for problem-solving and decision-making.

(*iv*) comprehensive understanding of remediation technologies; (*v*) the range of decisions that have to be made and their interrelationships which requires different decision-making techniques and decision support methods; *and* (*vi*) the different policy and regulatory requirements and their contexts.

The development of contaminated land management DSS also requires consideration of the evolution of DSS as the decision-making process changes due to changes in the underlying scientific and technical understanding of issues of land contamination, resulting in a change in policy requirements and ultimately the decision-making process. With different parts of DSS evolving at different times and in different ways, the developed DSS needs to make provisions for adapting the different parts as they independently evolve (Gachet and Hättenschwiler 2003). An effective DSS therefore needs to be developed to permit change quickly and easily (Sprague 1980). Taking into account these requirements for developing contaminated land management DSS, a framework for developing contaminated land management DSS is presented. The framework is based on: (*i*) sound principles of software engineering; (*ii*) characteristics of DSS development, use and evolution; *and* (*iii*) contaminated land management decision situation and decision-making requirements. The framework is generic enough to be applied for developing different types of contaminated land DSS at different management decision-making levels.

5.3 LIMITATIONS OF SOFTWARE DEVELOPMENT MODELS FOR THE DEVELOPMENT OF DECISION SUPPORT SYSTEMS

Many of the DSS reported for contaminated land have been developed and presented using the traditional *build and fix* approach to software development. This involves developing an initial system and reworking it as many times as necessary, until all the functionality required has been implemented. In this developmental approach, no attempt is made to formally specify DSS requirements or design its structure, and no provision is made for DSS evolution after it is operational. Many software development models exist which can be used to structure and formalise DSS development process. Although these differ in the degrees of structure they provide and developmental approach, all structured models involve these fundamental development phases (Sommerville 2001):

- Specification phase involving understanding and defining the software functionality and its operating constraints.
- Design phase involving building conceptual model(*s*) of the system.
- Development phase involving developing the software product from the conceptual models to meet the software specifications.
- Validation to ensure the software does what it is intended to do.
- Evolution to adapting software to meet changing needs. Maintenance costs tend to be higher than the cost of all the other phases put together, and good design is essential for significant reduction in maintenance time and costs.

One of the structured methods that can be used to develop DSS is the waterfall model, which is a highly structured method that breaks the software development life cycle into distinct phases (*Fig 5.1*). The waterfall model approach assumes the system requirements are fully known and understood and will not change over time, and requires each phase to be completed and documented before the next phase begins. However except for the very simple and small scale DSS that has well defined requirements that do not change over the development process, this approach may be too inflexible as it does not allow for iteration between the phases. The waterfall model also makes no provisions for software evolution, therefore if the decision-making process changes new DSS will have to be developed.

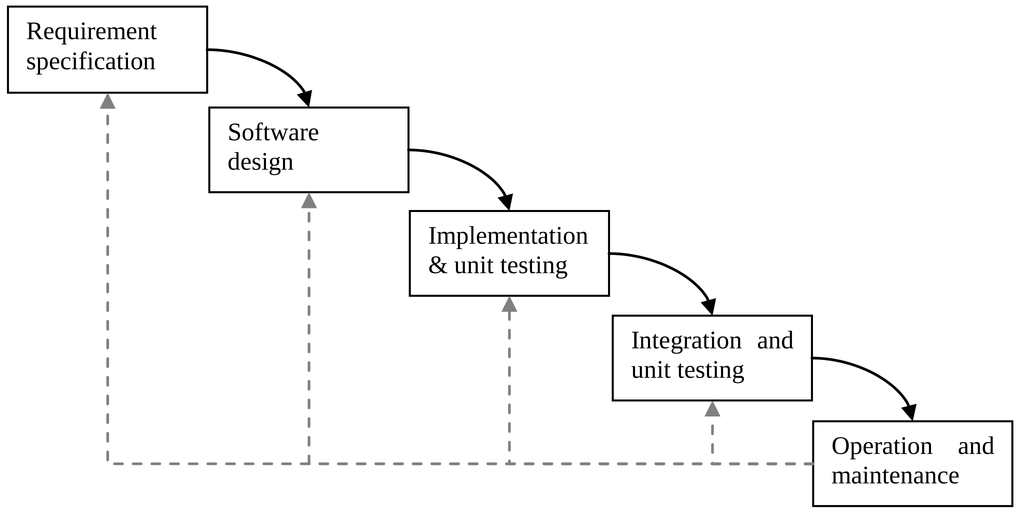

Figure 5.1 – Modified waterfall model reflecting necessary iteration in software development (after Royce 1970)

Another structured software development model that can be used to develop DSS is evolutionary prototyping, which is less structured than the waterfall model but more flexible. The evolutionary prototyping approach involves rapidly developing a DSS prototype from very abstract specifications by integrating the specification, development and validation phases (*Fig 5.2*). The prototype can then be refined with decision maker(s) input to develop other prototypes, until a satisfactory DSS is produced. The evolutionary prototyping approach allows decision makers' to adapt the DSS requirements as it is being developed. Although this may lead to DSS that is poorly structured and difficult to understand and maintain, the finished DSS can be re-implemented using a structured approach to produce a robust and maintainable system (Sommerville 2001).

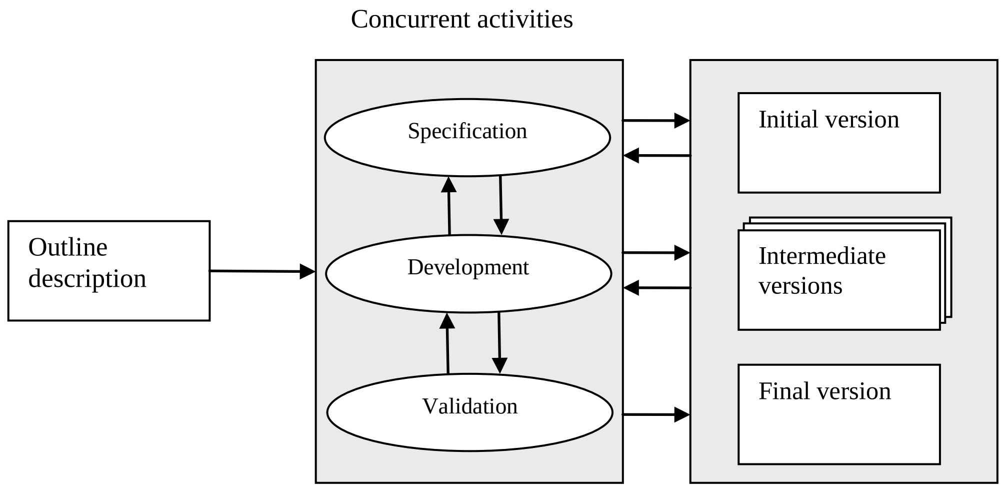

Figure 5.2 – Evolutionary prototyping software development model (Sommerville 2001)

Both the waterfall model and evolutionary prototyping model do not allow for any iteration in the development process. Other development models that are specifically designed for software evolution include the incremental development model (Mills et al 1980) and the spiral model (Boehm 1988). The incremental development model is based on the waterfall model, and allows for system functionality to be developed in increments (*Fig 5.3*). The process begins with defining outline specifications, and then incrementally developing the system from the most important to the least important until all specifications have been implemented. The incremental development model requires well defined User Interface (UI) because different functionalities of the software are developed at different times. Different development models may be used for developing different increments, such as the waterfall model for well defined increments or evolutionary prototyping for the less structured increments. For complex systems however, it may be difficult to map the right functionalities to the right increments, and to identify common functionalities that all increments require (Sommerville 2001).

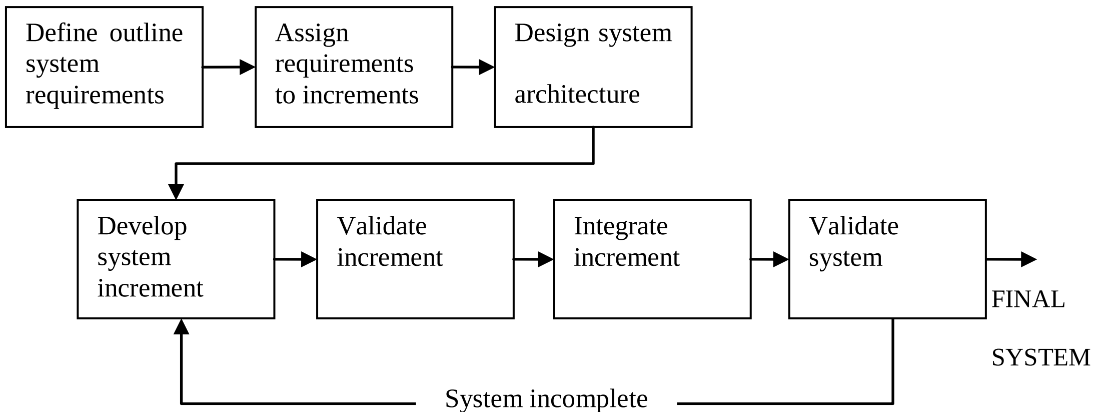

Figure 5.3 – Iterative software development model (Sommerville 2001)

The spiral approach models the software development process as a spiral rather than as a sequence of activities (*Fig 5.4*). This is a risk-based approach which accounts for risk(s) at each spiral loop. There are no fixed phases such as specification, development or validation, and each loop in the spiral can be undertaken using other development models such as the waterfall or evolutionary model. Each loop in the spiral represents a phase in the development process, and each phase involves four segments: (*i*) defining objectives and constraints of the loop; (*ii*) identifying the risks associated with activities within the loop, assigning priorities to them and taking steps to reduce those risks; (*iii*) development and validation of the prototype; *and* (*iv*) reviewing the loop and planning for the next phase, and if risk(s) cannot be resolved the development process is terminated.

It is evident that these traditional software development models have limitations for developing DSS because DSS evolve and need to be adapted after they are operational, at considerable time and effort. The waterfall approach is too structured and does not allow for software evolution, even during development. Evolutionary prototyping approach allows for iteration during development, although this advantage may lead to poorly structured software that is difficult to document and modify. Both of these models do not provide means for DSS to evolve over its operational life time. Even the models that support iteration do not allow for software evolution.

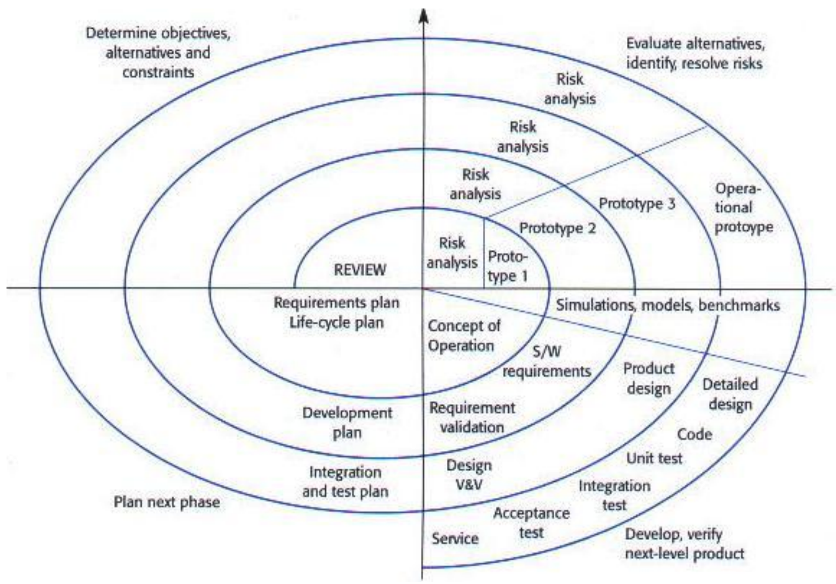

Figure 5.4 – The spiral software development model (Boehm 1988)

Given these limitations of traditional software development approaches for developing DSS, a Component Based Software Engineering (CBSE) (Szyperski 1997) approach to DSS development is proposed. The CBSE approach specifically address issues of software reuse, and involves developing parts of the DSS as independent components that can then be integrated (*composed*) into a DSS through a common interface. The key characteristics of components include (Szyperski 1997; Somemrville 2004):

- A component is an independent executable system, with the components functionality accessed using a well defined User Interface (UI);
- Components conforming to a standardised model that defines the component UI, meta-data, documentation, composition and deployment;
- Each component is capable of performing its functionality independent of the system, and the addition or removal of component(s) should not affect other components, functionality of other components; and
- Component implementations to be encapsulated (blackbox) as the functionality of component-based software should be wholly UI-driven.

5.4 COMPONENT-BASED APPROACH TO DEVELOPING DECISION SUPPORT SYSTEMS FOR CONTAMINATED LAND MANAGEMENT

The development of DSS requires consideration of the DSS application area, the functionality envisaged, technical complexity of the development process and decision-making process, decision-making level, decision maker(s) involved with the development and use of the DSS, and the approaches to software and hardware implementation (Black and Stockton 2009). The component-based DSS development life cycle involves: (*i*) specifying requirements by defining the DSS objectives and its functional and non-functional requirements; (*ii*) identifying components and their functionalities; (*iii*) designing the DSS architecture, which describes the structural properties of components and component interrelationships; (*iv*) independently developing and evaluating the DSS components; (*v*) composing the individual components into the DSS; *and* finally (*vi*) evaluating the DSS in a timely and cost effective way (*Fig 5.5*). These phases of the development framework are further described in the sub-sections that follow.

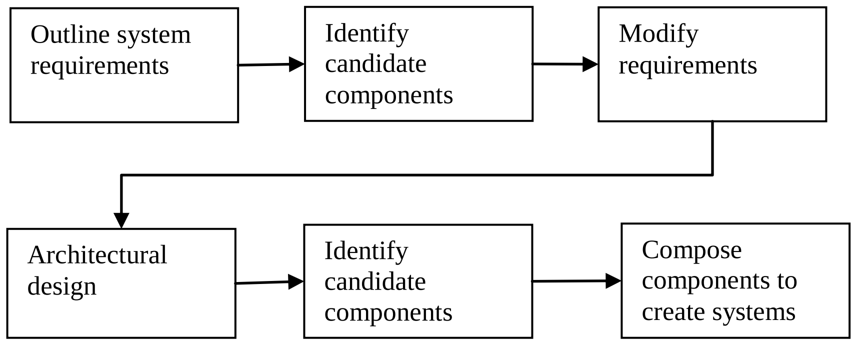

Figure 5.5 – Component-based software development model (Sommerville 2001)

5.4.1 Specifying the decision support system requirements

The first stage of the component-based DSS development process involves defining the DSS specifications by analysing the DSS objective(s), its functional and non-functional requirements, *and* the constraints on the DSS development, use and evolution. Defining DSS objectives includes defining the usage of the DSS; the type of decision outcome(s); *and* the decision-making level the DSS will support. The uses envisaged or desired for contaminated land management DSS include: (*i*) identifying realistic management choices; (*ii*) integrating information into a coherent framework for analysis and decision-making discerning key information and impacts decision-making from more basic information; (*iii*) providing a framework for transparency (*i.e.* all parameters, assumption, and data used to reach the decision are clearly documented); *and* (*iv*) ensuring that the decision-making process itself is documented (Bardos et al 2000).

Functional requirements are used for defining the DSS operational functionalities, which depend on the role the DSS will play in meeting the decision objective(s) and on technical objective(s) of the DSS (Black and Stockton 2009). These can be elicited through the decision maker(s), knowledge about the decision situation and the decision-making process and/or by evaluating similar existing systems. Non-functional requirements are used for describing the behaviour of the DSS in its operational environment and covers issues as broad as reliability of the DSS in providing accurate and timely support when needed, performance of the DSS, safety and security especially in cases of sensitive data. Non-functional requirements do not directly impact on functional requirements (Black and Stockton 2009), although they are often critical in software systems, with failure to achieve some minimal defined requirement making a system unusable (Sommerville 2001). Requirements specification is a particularly critical stage in the software development process, as errors not identified and dealt with at this stage could invariably lead to problems later on in the design and implementation stages.

5.4.2 Identifying components and their functionalities

Several architectures have been proposed for developing both generic and discipline specific DSS, including Sprague (1980), Bonczek et al (1981), Marakas (1999), Turban and Aronson (2001), Cortés et al (2002), Denzer (2005), Matthies et al (2007), Sànchez-Marrè et al (2008), *etc*. These architectures all include three common core components:

- a model component and a Model Management System (MMS),
- a database component and a Database Management System (DBMS), *and*
- a User Interface (UI) component.

The model component contains any model(s) required for supporting decision-making, which could include mathematical, numerical or statistical models for simulating contaminant behaviour and transport, Geographic Information System (GIS) for spatial analysis of contaminant locations and hotspots, and/or intelligent models. The modelling component must contain at minimum a Decision Analysis (DA) model such as Multi Criteria Decision Analysis (MCDA) model for decision support. The MMS is used to manage the interactions between models, and for interfacing the model component with the other DSS components. The database component contains database(s) and/or data warehouse(s) for storing, managing and/or retrieving information for decision support. The DBMS is used for manipulating the databases, and for interfacing the database component with the other DSS components. The UI component is used by the decision maker(s) to interact with the DSS. Much of the power, flexibility and usability of DSS are derived from the capabilities of the UI (Sprague 1980); and for many users, the UI is the DSS which makes it the most important component as it heavily influences how the DSS is perceived and used (Power and Sharda 2009).

These core components can be modified for specific DSS applications, and several discipline specific and problem specific architectures have been proposed. Examples of discipline specific architectures include Cortés et al (2000), Denzer (2005), Matthies et al (2007), Poch et al (2003) and Sànchez-Marrè et al (2008) for EDSS. No problem specific architecture for developing contaminated land management DSS currently exist. However given that contaminated land management DSS are a subtype of EDSS, the architecture of EDSS can be modified to contaminated land management DSS. Despite many of the EDSS reported having only one of four components: (*i*) a model component; (*ii*) GIS component;

(*iii*) decision support component; *or* (*iv*) a data management system; an EDSS should consist of at least two of these four components, the second of which should be a good data management system (Denzer 2005). Although one may argue that a data management system could be considered DSS, today's consensus is that EDSSs must adopt a knowledge-based (*modelling*) approach (Poch et al 2003). Ideally an EDSS should consist of an additional environmental modelling component, such as numerical or statistical simulation model(*s*) or GIS for spatial analysis (Fedra 2000).

5.4.3 Architectures of decision support systems

Software architecture is used to describe the overall structural properties of the software components, the relationships between the components, and their interrelationships; *and* the principles and guidelines governing their design and evolution over time (Bass et al 2003). An important aspect of software architecture is not just the architecture itself, but the rationale for why it is designed that way. Consideration must therefore be made to ensure the architectural design process is well documented (Eeles 2006). DSS architecture is therefore the blueprint for component integration. Although it is widely accepted that DSS should consist of at least a database, model and UI components, not much research has been made into how these components should be integrated and the relationships between them. Developing software architecture is an essential part of DSS development process for: (*i*) making early design decisions on development, deployment and maintenance of the software; (*ii*) communicating software requirements with stakeholders using a common abstraction; *and* (*iii*) providing transferrable abstraction of the system by supporting reuse of the DSS components at the architectural level; *and* (*iv*) composition of externally developed components (Bass et al 2003). The architectural design process involves designing the structure of the individual components showing the relationships between components and that of the integrated system (*Fig 5.6*).

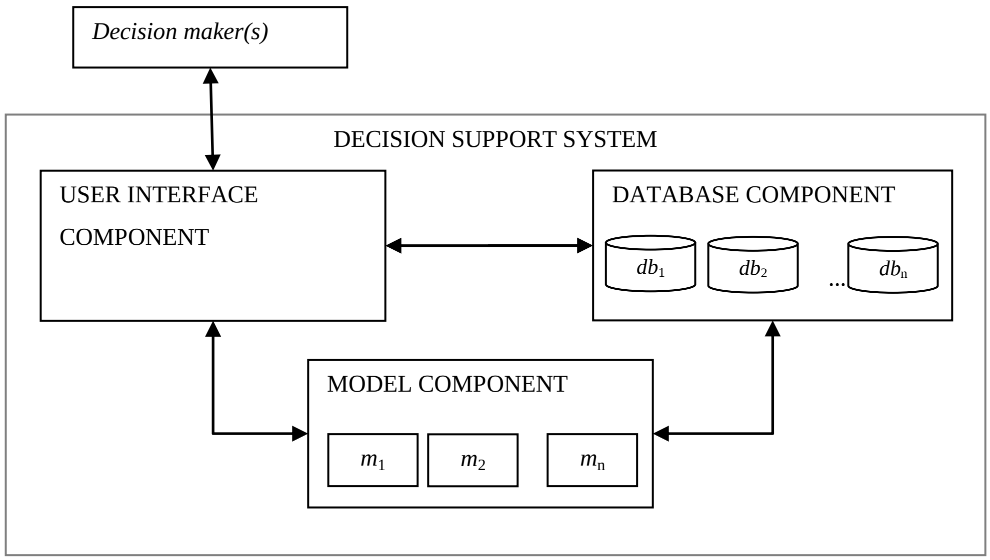

Figure 5.6 – Architectural representation of component-based decision support systems

The DSS architecture also includes the infrastructure the DSS is to be deployed on. Most DSS are deployed on two main infrastructures, as executable programs on individual computers, or on a distributed infrastructure. Distributed DSS are deployed over the Web, which often use *n*-tier client-server architecture (*Fig 5.7*). Client-server architectures are a powerful infrastructure for distributed or collaborative decision-making, and are useful for supporting distributed and group decision-making, and for information sharing. There are three types of client-server model: (*i*) static model using basic HTML[33]; (*ii*) client-side

[33] HTML – HyperText Markup Language

processing using CGI[34]; *and* (*iii*) server-side processing using scripting languages. Client side with processing is done locally on the user computer, and server side processing done on a centralised server. Most recent DSS are developed and deployed on client-server architecture with server-side processing.

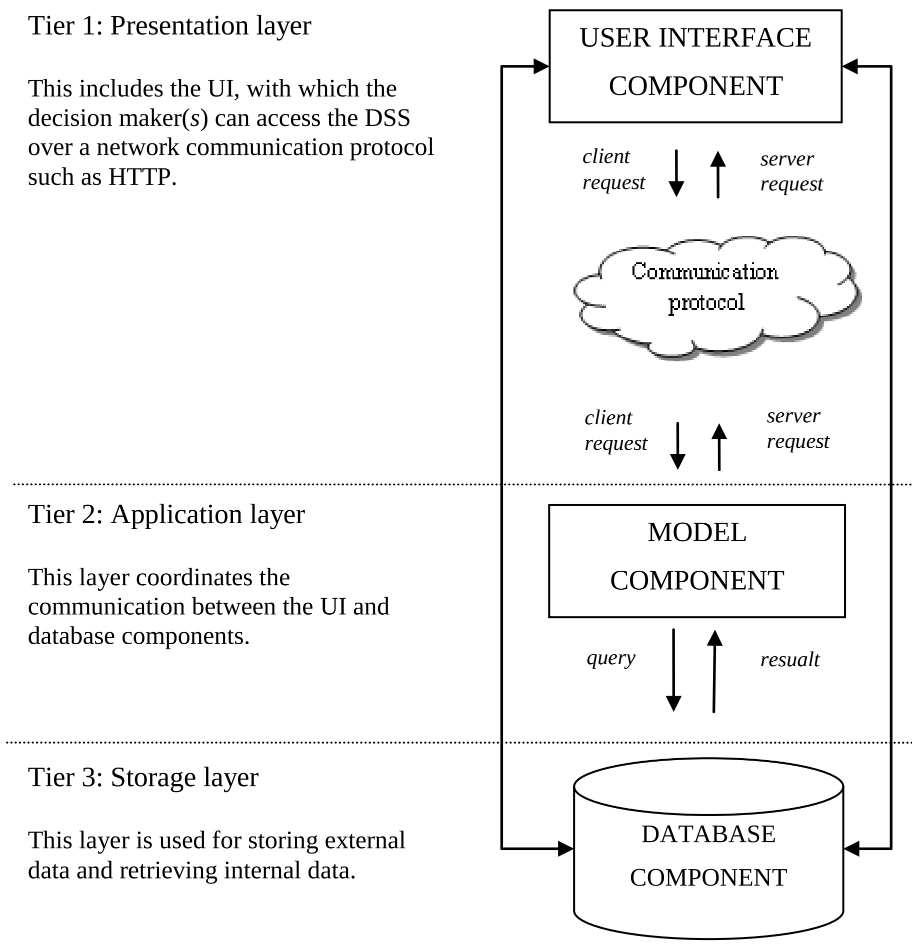

Figure 5.7 – Three-tier client server architecture for distributed DSS

5.4.4 Implementation of the decision support systems

There are two main approaches to software development - open and closed source. Closed

[34] CGI – Common Gateway Interface

source development usually involves commercial and proprietary technologies where software is developed as a blackbox, with limited modification or distribution of the end product. Even when distribution is allowed, a closed source development environment will be needed by the recipients to view the software. Open source development involves developing software with the software code freely available without cost or obligation, under varying licensing conditions. There are many advantages to using open source development approach, especially in research environments, which includes viewing, modifying and distributing software at no cost.

In the case of a component-based DSS, open source development will give flexibility in integrating more models or databases to their respective components, or with developing other components and adding them to the DSS. The developed DSS should be transparent, standardised and robust. Transparency is achieved by writing standardised and well documented code that other programmers can understand. Robustness requires DSS to perform well outside of the developers assumptions. In the case of component-based DSS, when one component fails, it should not affect other components or the system itself. The advantage of component-based DSS development is with allowing for the integration of components developed on different platforms and architectures.

Language independence allows different components to be developed using mixed language programming, because the end product of each component is essentially a blackbox to the other components. Platform independence is good practice in software development as it allows the end product to be used on any computer system, independent of its operating system. Different components of the DSS will require different developmental approach methods, and techniques, and will therefore need to be developed separately. The development of DSS will therefore include the independently developing the database, model, and the UI components.

There are two approaches to data representation in the database component – flat files or relational databases. Flat file databases store all the information of the database in one table, with each record stored per line. Flat file databases are often saved as *.txt*, *.csv* or *.xml* file types which can be accessed with the simplest text editors. This is inadequate for complex databases however, because flat file databases are prone to data duplication and there is no automation of database manipulation. Relational databases contain multiple related tables, and provide many functionalities such as searching, or joining of records. Different relational databases exist, which could include proprietary (*e.g*. Microsoft Access, Microsoft SQL, Oracle) and Open Source (*e.g*. OpenOffice Base, PostgreSQL, MySQL) alternatives. The same advantages and limitations of Open Source and commercial/proprietary software development apply with DBMS.

The model component can include different models such as mathematical and numerical models, decision-making models, and/or intelligent models such as fuzzy, knowledge-base or neural networks. Different model types will require entirely different modelling and development paradigms. For example, contaminant behaviour and transport problems are best formulated using partial differential equations, solved using finite difference/element models and programmed using the FORTRAN programming language; spatial modelling often involves either data visualisation or statistical analyses; and management decision-making processes are best modelled using Multi Criteria Decision Analysis (MCDA) methods. An obvious advantage of using a common UI in component-based DSS development approach is with integrating the development paradigms.

Different programming languages such as (*e.g*. Java, C++, Visual Basic), or a combination of scripting languages (*e.g.* ASP[35], JavaScript, PHP[36] Perl or Python), markup languages (*e.g*. HTML, XHTML[37], XML[38]) and styling languages (*e.g*. CSS[39]) are used for UI development. DSS code written using programming languages need to be compiled before use, but once compiled can be re-used any number of times. Codes written in scripting languages are compiled each time the DSS is used. Compilation involves translating the DSS code from machine language into executable programs. Markup languages are used for structurally annotating (marking up) text using markup elements in <*tags*>. Combinations of the different programming paradigms are often packaged as comprehensive and integrated bundles for software development.

A well developed and widely used example of this is the *Linux/Windows, Apache, MySQL and Perl/PHP* (LAMP/WAMP) bundle which includes the development operating system, server, database and a scripting language respectively and is used with markup language(*s*) for developing powerful, open source and secure Web applications. Another increasingly used software development bundle is the *Asynchronous JavaScript and XML* (AJAX) which is also used alongside markup language(*s*) for developing dynamic Web applications. The final stage of the component-based DSS development process is

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[35] ASP – Active Server Pages
[36] PHP – Hypertext Preprocessor
[37] XHTML – eXtensible HyperText Markup Language
[38] XML – eXtensible Markup Language
[39] CSS – Cascading Style Sheets

component integration, which involves composing the independently developed and independently functional components into an overall DSS.

5.4.5 Evaluation of decision support systems

An important part of the development of DSS is evaluation in order to establish what the DSS knows, knows correctly, and/or what it does not know (O'Leary 1987). The DSS evaluation process typically involves: (*i*) verification, validation and quality control of the usability of the overall system; *and* (*ii*) investigating the assumptions and limitations of the DSS, its appropriate uses and why it produces the results it does (Borenstein 1998). Verification involves testing and debugging the software code, and is typically carried out throughout the development process. Validation is involves testing the appropriateness of the DSS in supporting real world decision problems. Since it is impossible to prove a DSS is a truthful representation of the real world, validation is primarily concerned with demonstrating that the DSS has appropriate underlying relationships to permit an acceptable representation (Finlay et al 1988).

The validation of DSS is as critical as its development to ensure adequate performance in real world applications, yet few works are devoted to this aspect of DSS development (Sànchez-Marrè et al 2008). Research has found little validation is carried out during or after development of DSS (Finlay 1988, Sailors 1996), with most evaluation effort preferentially directed at verification to the detriment of validation (Mosqueira-Rey and Bonillo 2000). The two methods of validating component-based DSS are: (*i*) validating individual components; *and* (*ii*) validating that an acceptable output is achieved for different sets of decision problems. A combination of both methods, *i.e.* independently validating each component *and* validating the overall DSS is necessary for validating complex systems (Finlay 1988). The validation process can be carried out by either: (*i*) functional testing, which involves testing DSS inputs and outputs, usually against real life case studies; *and/or* (*ii*) structural testing, which involves testing the design and development of the individual components of the DSS (Sailors et al 1996).

Most of the DSS reported for contaminated land management have at the minimum been functionally validated against real case studies. Each case study is different however, and as a result functional testing on its own often only evaluates the functionalities needed for solving particular case studies, and may not evaluate all aspects of the DSS or in cases some of the DSS component(s) at all (Finlay 1988, Sailors et al 1996). Although functional testing on its own can be effectively used to test complex systems, it is not adequate for debugging. Structural testing in isolation is also insufficient for testing the interactions between individual elements, and therefore neither functional nor structural testing in isolation are adequate for validating complex system.

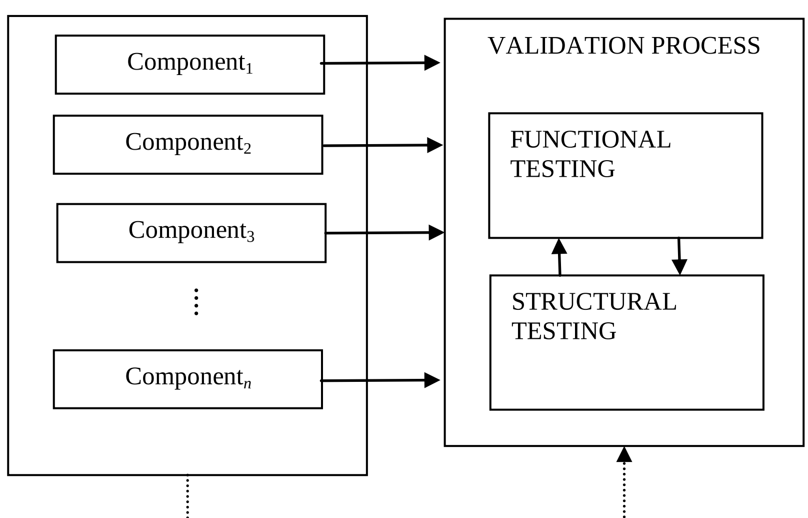

Figure 5.8 – Validation process of component-based DSS (after Finlay 1988 and Sailors et al 1996)

Effective DSS validation will therefore need to include both functional and structural testing (Sailors et al 1996) using well defined validation criteria (Mosqueira-Rey and Bonillo 2000). An appropriate evaluating approach for component-based DSS should therefore include: (*i*) verification throughout the DSS development process by testing and debugging the source code to ensure there are no errors in the DSS code; (*ii*) validation of each component independently both functionally and structurally, *and* (*iii*) validation of the integrated DSS both functionally and structurally (*Fig 5.8*).

5.5 CONCLUSION

Many frameworks for developing both generic and discipline specific DSS have been proposed. However no single framework dominates and the development of DSS is still largely an *ad hoc* process. This is mainly because the development of the DSS is a multidisciplinary process involving knowledge of the DSS application area and techniques and tools from a wide range of disciplines. This chapter presented a framework for the development of contaminated land management DSS, taking into account: (*i*) contaminated land management decision-making process and its constraints: (*ii*) the underlying multidisciplinary information required for contaminated land management decision-making; (*iii*) the range of management decisions that can be made; (*iv*) the different policy contexts; *and* (*v*) evolution of the DSS resulting from the changes in the underlying scientific and technical understanding of land contamination. The framework is based on the component-based approach to software development, which explicitly addresses issues of re-usability by developing different parts of the DSS as independent components.

6 DEVELOPMENT OF DECISION SUPPORT SYSTEM FOR CONTAMINATED LAND MANAGEMENT

6.1 INTRODUCTION

Using the framework described in the previous chapter, a Decision Support System (DSS) for the sustainable management of petroleum hydrocarbon contamination has been developed. As explained in chapter 1 a *problem specific* DSS was developed. Problem specific DSS are tailored for specific problems, which can be applied to different locations with the same problem domain (Rizzoli and Young 1997). These have many advantages over problem and location specific DSS, which are developed for a particular problem and location, and generic DSS for different types of problems, but cover much lower detail and provide generic solutions to complex management problems the most important of which is their ability to automate the solution of similar problems.

A number of DSS have been developed for the integrated management of contaminated sites, many of which have been reviewed in chapter 4. These include: ERA-MANIA – a DSS for ecological risk assessment (Critto et al 2007); DESYRE – a spatial DSS for integrated management of contaminated mega-sites (Carlon et al 2007); SADA – a spatial DSS for ecological risk-based remediation design (Purucker et al 2009); and DST – a decision support tool for the prioritisation of risk management options for contaminated sites (Sovari and Seppälä 2010). None of these DSS can be applied within the context of the current UK contaminated land regime however, and none of these explicitly addresses human health issues from land contamination. With respect to remediation design and options appraisal, none of these DSS explicitly addresses the sustainability of the remediation technologies. This thesis presents the development developed an integrated DSS for the sustainable management of contaminated land within the remit of the current UK contaminated land regime.

The remainder of this chapter presents the development of the DSS. Section 6.2 provides a background to petroleum hydrocarbon contamination, its occurrence and effects on human health, and the UK approach to managing it. Section 6.3 presents the design of the DSS. Section 6.4 presents the development of the DSS database. This is covered in detail in Appendix IV. Section 6.5 presents the development of the DSS decision model. The development of the decision model is covered in detail in Appendix V. Section 6.6 covers the development and operation of the DSS User Interface (UI). The developed DSS consists of three key phases covering the overall management decision-making process: (*i*) preliminary qualitative risk assessment which is used for identifying the relevant legislation and regulatory enforcement that will apply at a site based on different input parameters; (*ii*) generic quantitative risk assessment which involves comparing measured site sample concentrations with Generic Assessment Criteria (GAC) values; *and* (*iii*) options appraisal of remediation technologies and remediation design, which involves the sustainability appraisal of selected remediation technologies.

6.2 PETROLEUM HYDROCARBON CONTAMINATION

Petroleum hydrocarbon contaminants are amongst the most commonly occurring at contaminated sites. According to the *European Environment Agency* (EEA) approximately 14.1 percent of identified contaminated lands in its member countries are caused by the oil industry, with heavy metals, mineral oils and hydrocarbon contaminants constituting approximately 90 percent of the total contaminants found on sites (EEA 2007). The most commonly occurring of which are metals and inorganic compounds in England and Wales are (*Fig 6.1*), with hydrocarbons specifically being amongst the most commonly occurring contaminants found in groundwater (*Fig 6.2*). Petroleum hydrocarbon contaminants in soils pose potentially unacceptable risks to human health either directly through the food chain and water supply, or through inhalation of dust or dermal contact.

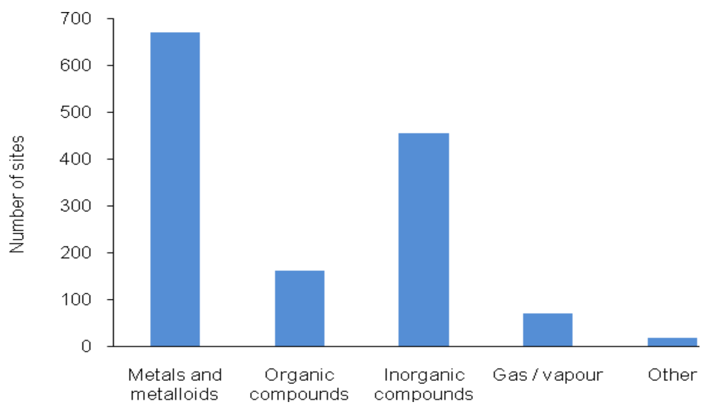

Figure 6.1 – Main contaminants reported in contaminated land sites in England and Wales[40], 2007(EA 2008)

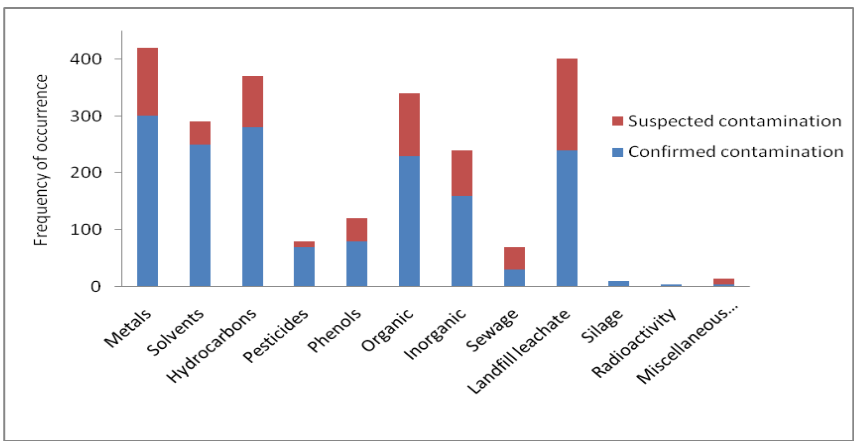

Figure 6.2 – The frequency of occurrence of contaminants in groundwater in England and Wales (UK Groundwater Forum c2010)

[40] Note: More than one contaminant can occur at an individual site.

6.2.1 Technical overview

Petroleum hydrocarbon, also commonly known as Total Petroleum Hydrocarbon (TPH), contamination occurs as a result of the processing, storage and use of crude oil based products. The processing of crude oil involves distillation and separation using a variety of chemical and physical processes into different products for different uses (*e.g*. petrol, kerosene, diesel, aviation fuel, *etc*) and into different hydrocarbon fractions, containing major hydrocarbon compounds. These compounds provide essential resources for energy, transportation, agricultural feed stock and the synthesis of plastics (Russell et al 2009). The hydrocarbon compounds have different physical and chemical properties and as such behave differently once released into the environment. TPH contamination is caused by the release of crude oil based products into the environment.

Petroleum products are complex mixtures of hydrocarbon fractions derived from blending products obtained from the processed crude oil with brand-specific performance enhancing additives (HPA 2007). Although hydrocarbons are simple organic compounds mostly comprising of only carbon and hydrogen, TPHs have complex mixtures each of which may contain hundreds of individual chemicals that are closely related yet each with its own toxicological properties. The fractions are characterised according to their boiling point ranges, and grouped according to their fate and transport properties. The type of crude oil, the way it is processed, and its use and behaviour once released in the environment results in hydrocarbon residues of extreme chemical complexities.

Petroleum itself is not particularly toxic, and accidental poisoning is very rare (HPA 2007). However its release into the environment through industrial releases, spills, leaks, aerial deposition or as by-products of other processes causes severe contamination of the environment (*Fig 6.3*). Physical, chemical and biological processes weather the chemicals over time, which causes changes in composition and complexity of the TPH contaminants, potentially changing the intensity and significance of the risk(*s*) posed (EA 2005). The weathering of the mixtures results in the partitioning into different solid, liquid and gas phases. Physical, chemical and/or biological processes also affect the location(*s*) and concentration(*s*) of hydrocarbons at any particular site (ATSDR 1999). This significantly affect known potential effects, posing significant potential risks to human health, including organ damage, toxic, mutagenic and carcinogenic effects (HPA 2007).

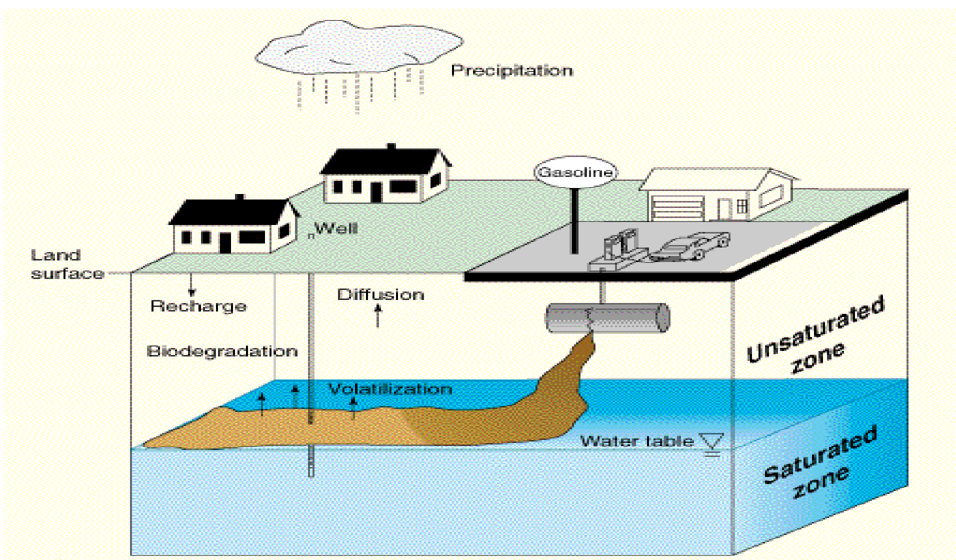

Figure 6.3 – Conceptual model of the fate and transport of petroleum hydrocarbons in the sub-surface environment (USGS 1998)

The human health effects of petroleum hydrocarbons depend on several factors, including the amount to exposure, the exposure route, exposure duration, and the type(*s*) of the contaminant(*s*) at any particular site. Exposure is defined as the amount of a chemical that is available for intake by a target population at a particular site, and is quantified as the concentration of the chemical in the medium (*e.g.* air, water, food) integrated over the duration of exposure and expressed in terms of mass of substance per unit mass of soil, unit volume of air or unit volume of water (mg kg^{-1}, mg m^{-3} *or* mg L^{-1} respectively).

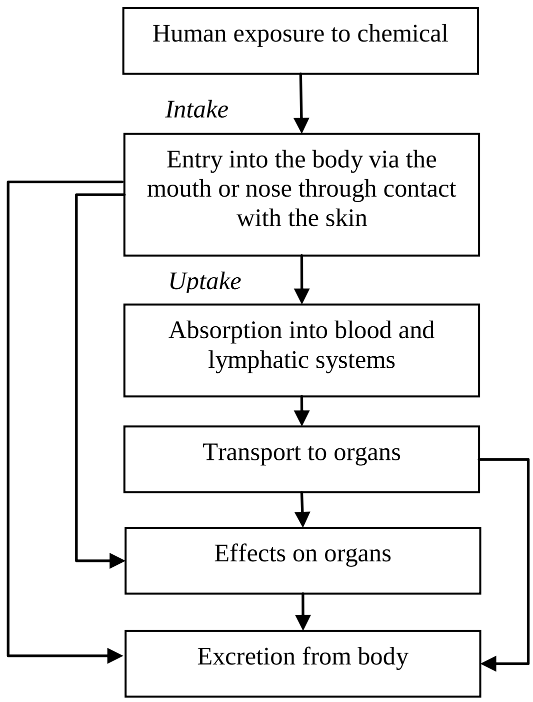

Figure 6.4 – Relationship between exposure, intake and uptake (DEFRA 2001)

Exposure that occurs externally is referred to as the *intake dose* (intake), and is defined as the amount of chemical entering or contacting the human body at the point of entry by ingestion, inhalation or skin contact. Actual intake is a function of the chemical characteristics and the nature of the target population and their behaviour patterns. Intake is expressed in terms of mass of substance per *kg* body weight over a period of time (mg kg^{-1} bw d^{-1}). In many cases there is no distinction made between the intake of contaminants that are bound to soil and those which occur as a vapour or are released during processes like digestion into solution. *Uptake dose* (uptake) is the amount of a contaminant that reaches the circulating blood having been absorbed by the body through the skin, the gastrointestinal system and the pulmonary system, and is expressed in terms of mass of substance per unit volume of blood (mg L^{-1}). The relationship between the terms exposure, intake dose and uptake dose, is illustrated in *Figure 6.4* (DEFRA 2001).

6.2.2 Risk assessment of petroleum hydrocarbon contamination

The UK *Environment Agency* (EA) has published a comprehensive technical framework for the evaluation of human health risks from petroleum hydrocarbon contamination in soils. The framework is within the context of the current contaminated land regime in Part IIA EPA, and is in line with the UK tiered risk-based approach to contaminated land management. The framework uses a combination of indicator compounds and petroleum hydrocarbon fractions which are based on fate and transport properties. The basis of the framework is a combination of the US *Total Petroleum Hydrocarbon Criteria Working Group* (TPHCWG) approach and the *American Petroleum Institute* (API) method for evaluating human health risks from TPH modified to fit with the UK context. The combination of the two frameworks was necessary because the TPHCWG approach is suitable for refined products like petrol and diesel, but not suitable for heavier fractions which the API method addresses.

The framework has published detailed toxicological (TOX) reports based on three generic land use scenarios and the likely exposure pathways. The relative contributions of each pathway to overall exposure are modelled based on the fate and transport properties of the hydrocarbon fractions. The land use scenarios are for (*i*) residential; (*ii*) allotment; *and* (*iii*) commercial/industrial land uses (EA 2005), which describe the type of people with access to a site and how they might potentially behave. The framework has developed *Soil Guideline Values*[41] (SGVs) as intervention values for assessment of risks in relation to land use that are representative of a range of generic site conditions taking into account studies of social behaviour (DEFRA and EA 2002*d*), which are used as *Generic Assessment Criteria* (GAC) values in risk assessment. The SGVs are derived from the CLEA model, which uses the toxicity of soil contaminants with estimates of potential long-term exposure by adults and children for the three land use scenarios (DEFRA and EA 2002*c*).

SGVs in the CLEA model only apply to assessment of direct human exposure to soil contaminants, and do not consider other receptors such as controlled waters, ecological systems, property or the health and safety of workers. The framework does not address issues of aesthetics such as odours and staining however, even though aesthetics represent a key driver for remediation and redevelopment of petroleum hydrocarbon contaminated sites. This is because aesthetic issues vary from site to site, making it difficult to develop generic guidance, and aesthetics are outside the scope of risk-based assessment of human health risks (EA 2005). The framework has also not published *Health Criteria Values*[42] (HCVs) for the derivation of the SGVs, but describes how to derive them for both indicator compounds and hydrocarbon fractions (DEFRA and EA 2001).

Indicator compounds are often the key risk drivers at petroleum hydrocarbon contaminated

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[41] Soil Guideline Values (SGV) are scientifically based generic assessment criteria that can be used to simplify the assessment of human health risks arising from long-term and on-site exposure to chemical contamination in soil (EA 2009).

[42] Health Criteria Values (HCV) represent the toxicological benchmark against which human exposure to soil contaminants is ultimately compared (2002*d*).

sites, and include both *threshold* and *non-threshold* contaminants[43] (*Table 6.1*). Indicator compounds represent the most toxic (*threshold*) and the most frequently occurring contaminants. Hydrocarbon fractions represent contaminants that only pose a risk after a certain exposure threshold has been exceeded (*non-threshold*). The assessment of fractions provides a more representative picture of risk at sites where the origin of the petroleum contamination may be unclear. Hydrocarbon fractions are grouped on the basis of their fate and transport properties, which are closely related to compound mobility in the environment and are based on their EC numbers.

Hydrocarbon fractions are grouped according to their structures into aliphatic and aromatic fractions as they have different properties, with aromatic fractions being more soluble in water and less volatile than aliphatic fractions with similar EC numbers. Hydrocarbon fractions are divided into 13 fractions, with leaching and volatilisation factors that differ by approximately one order of magnitude. The UK framework has a total of 16 fractions, with a split for aromatic and aliphatic fractions covering > EC35 to EC44, and a new combined aliphatic and aromatic fraction > EC44 to EC70 (*Table 6.2*). As fractions are characterised in terms of their boiling point ranges, each fraction may contain hundreds of individual chemicals that are closely related yet each with its own toxicological properties. The threshold toxicity of each of the 13 fractions is represented by a *reference dose* (RfD) and

[43] With toxicity, it is assumed that there is a threshold level of toxicant that needs to be present to produce an effect leading to adverse effects. With some hydrocarbons however, specifically for mutagenic and genotoxic carcinogens, there is no theoretical reason why a single molecular exposure should not result in a tumour or mutation, possibly expressed in subsequent generations. For these substances, no threshold can be assumed and it is accepted that they carry some risk at any slight level of exposure (DEFRA 2001). Genetoxic carcinogens are cancer-causing agents that can alter *deoxyribonucleic acid* (DNA) molecules.

reference concentration (RfC) for surrogate compounds[44] or mixtures.

Table 6.1 – Petroleum hydrocarbon indicator compounds in the UK framework for human health risk assessment (EA 2005)

NON-THRESHOLD INDICATOR COMPOUNDS	THRESHOLD INDICATOR COMPOUNDS
benzene [a, b]	toluene [a, b]
benzo(*a*)pyrene [a]	ethylbenzene [a, b]
benz(*a*)anthracene	xylene [a, b]
benzo(*b*)fluoranthene	naphthalene [a]
benzo(*k*)fluoranthene	fluoranthene
chrysene	phenanthrene
dibenz(*a*,*h*)anthracene	pyrene
indeno(1,2,3-*c*,*d*)pyrene	

a = published under the old CLEA model
b = published under the new CLEA model

Table 6.2 – Petroleum hydrocarbon fractions for use in UK framework for human health risk assessment, based on equivalent carbon number (EA 2005)

ALIPHATIC FRACTIONS	AROMATIC FRACTIONS
> 5 – 6	> 5 – 7
> 6 – 8	> 7 – 8
> 8 – 10	> 8 – 10
> 10 – 12	> 10 – 12
> 12 – 16	> 12 – 16
> 16 – 35	> 21 – 35
> 35 – 44	> 35 – 44
> 44-70	

[44] A surrogate is an individual compound or mixture within each fraction deemed to represent the toxicity of the fraction.

6.2.3 Management of petroleum hydrocarbon contamination

The management of TPH contamination is carried out within the existing risk-based framework for the management of contaminated land, with emphasis on a tiered risk-based assessment and decision-making approach (*Fig 6.5*). The management process begins with preliminary (qualitative) risk assessment which involves desk study, site reconnaissance, and collection of samples for chemical analysis. If there are any potential risk(*s*) present, the assessment process proceeds to the next tier, generic quantitative risk assessment, which involves comparing measured site sample concentrations with SGVs as GAC values. Depending on site specific circumstances, the exceedance of SGVs could indicate a Significant Possibility of Significant Harm (SPOSH) occurring. Assessment can proceed to either detailed quantitative assessment and/or remediation. In cases of uncertainty on whether SPOSH exists, detailed quantitative risk assessment is carried out. This involves the use of more detailed site specific information and criteria to calculate site specific assessment criteria using clearly defined algorithms (DEFRA and EA 2002*c*).

The effective remediation of petroleum hydrocarbon contaminated sites requires an understanding of all the processes involved; the properties of the product(*s*) and their behaviour in the environment; the effects and implications on potential receptors and a good understanding of the potential exposures and the implications on human health. Scientifically sound and practically feasible decisions need to be made to ensure that these effects are dealt with responsibly and effectively. Whilst on some sites the application of published guidelines can result in a satisfactory outcome, for many sites the nature and extent of the contamination present make clean up difficult and uncertain, particularly where sensitive land end uses are proposed (Nadebaum et al 2000). Effective management cannot be achieved on the basis of scientific concepts alone therefore, and expert judgement about the relative importance of different kinds of risk and about balancing trade-offs play an important role in the process (Shershakov 2009).

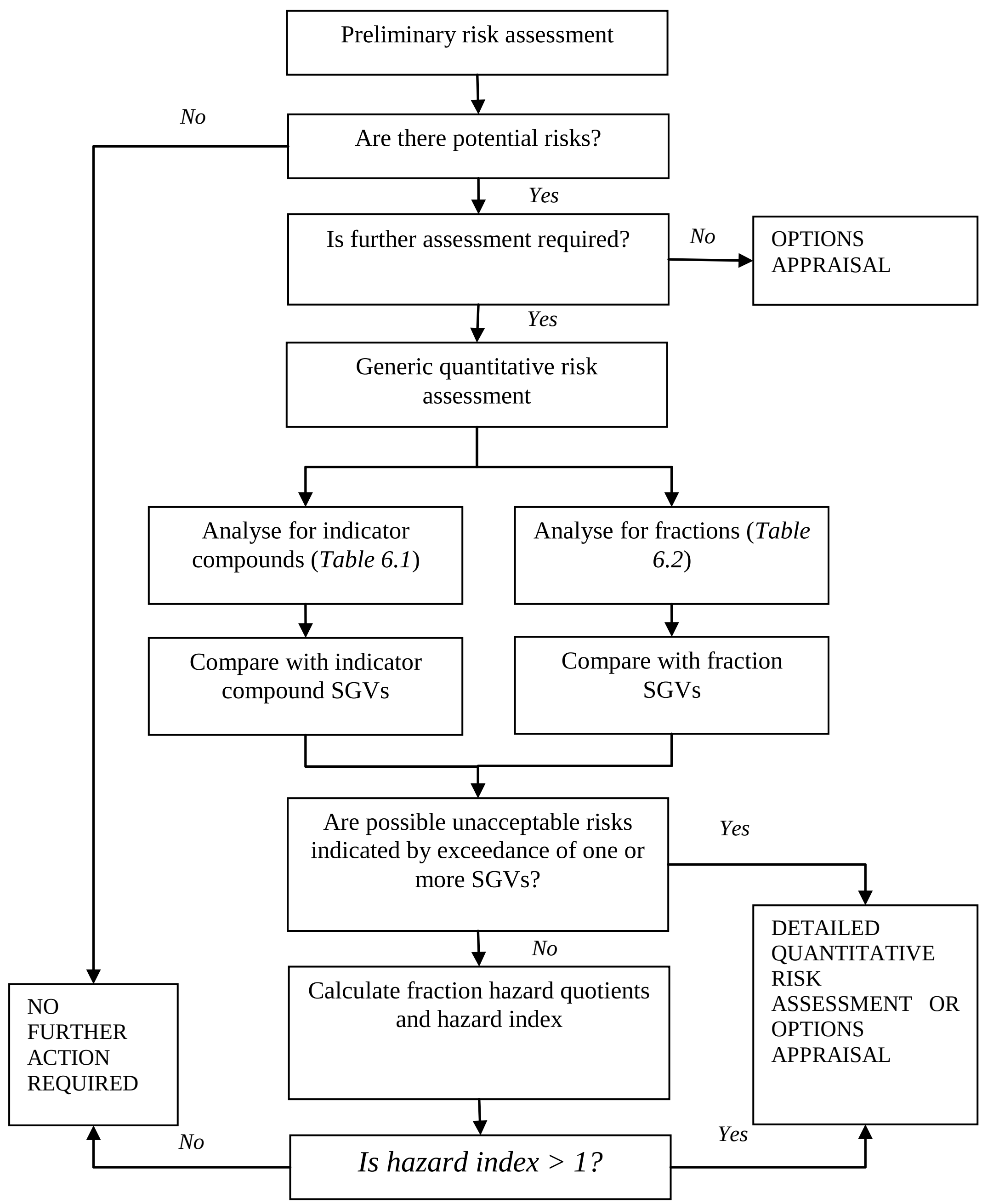

Figure 6.5 – UK framework for risk assessment of human health from petroleum hydrocarbons in soils (EA 2005)

6.3 THE DESIGN OF DECISION SUPPORT SYSTEM FOR SUSTAINABLE MANAGEMENT OF PETROLEUM HYDROCARBON CONTAMINATION

The remainder of this chapter presents the development of the DSS, which is based on the current UK contaminated land regime in Part IIA EPA, with supporting guidance from technical reports and professional expertise. The professional expertise was provided by *Exeter Environmental Consulting Services* (EECS), a small company which offer a wide range of geo-environmental consultancy services with emphasis on land development and waste management, whose staff have combined expertise of over 30 years. The developmental approach used is a component-based software development approach presented in chapter 5, using a mixed language programming approach for developing the different components (*Table 6.3*). Different components have different architectures, and developmental approaches which require different techniques and languages.

Table 6.3 – Technologies and programming language used for developing the decision support system

TECHNOLOGY	TYPE	VERSION
Linux (openSUSE)	Operating System	10.3
Apache	Web server	2.2.11
MySQL	Database server	5.1.36
PHP	Server-side scripting language	5.3.0
HTML	Markup language	4.0
JavaScript	Client-side scripting language	1.8.2
AJAX	Client-side scripting package	1.4.2
Cascading Style Sheet	Styling language	2.1
CLIPS	Expert system shell	6.30

6.3.1 Requirements specification

The main objective of the DSS is the sustainable management of contaminated land. Due to time limitation, the DSS is capable of managing only one type of contamination, petroleum hydrocarbon contamination. However, the DSS has been developed in a way that it can be easily extended to include other types of contamination. The management process involves: (*i*) preliminary (qualitative) risk assessment involving site characterisation based on site specific information (geology, hydrogeology, *etc*), desk survey, historic and future site use; (*ii*) generic quantitative risk assessment involving comparing contaminant concentrations with predefined *Generic Assessment Criteria* (GAC); (*iii*) appraisal of remediation technologies involving the selection and ranking of appropriate remediation technologies based on sets of sustainability criteria comprising economic, environmental and social criteria; *and* finally (*iv*) generation of risk assessment and management strategy reports. The overall objective of the DSS is to balance inevitable trade-offs between alternative options by minimising the overall management costs, minimising the environmental impacts from remediation, and maximising social benefits. This can be expressed by the following objective functions (Mantoglou and Kourakos 2007):

$$\min\ f^1(x_1, x_2, \ldots, x_n)$$

$$\min\ f^2(x_1, x_2, \ldots, x_n)$$

$$\max\ f^3(x_1, x_2, \ldots, x_n)$$

Subject to constraints:

$$a_i \le x_i \le b_i;\ \ i = 1,2,..n$$

$$g^j(x_1, x_2, ..x_n) \le c_j, \quad j = 1,2,...,k$$

where:

f^1, f^2, f^3 are the objective functions

$x, x_2, ..., x_n$ the decision variables

$g^1, g^2, ... g^k$ the constraint functions

$a_i \le x_i \le b_i, \ i = 1,2,..n$ the limits on the decision variables

$g^j(x_1, x_2, ..x_n) \le c_j, \ j = 1,2,...,k$ the limits on the constraints

6.3.2 Components of the decision support system and their functionalities

Contaminated land management DSS are a type of *Environmental Decision Support Systems* (EDSSs), which should ideally contain at least two of four components: (*i*) a model component; (*ii*) *Geographical Information System* (GIS) component; (*iii*) decision support component; *and*/*or* (*iv*) a data management component, the second of which should be a good data management system (Denzer 2005). Many of the EDSS reported in the literature have only one of four components, majority of which are a modelling and/or GIS component. Standalone GIS components are technically modelling components. Additionally, although one may argue that a data management system could be considered DSS, today's consensus is that EDSSs must adopt a knowledge-based approach (Poch et al 2003), and EDSS should also consist of an additional environmental modelling component, such as numerical or statistical simulation model(*s*) or GIS for spatial analysis (Fedra 2000). The developed DSS has three core components:

- A database component representing the DSS database and data management. The database component contains database for storing, managing and/or retrieving information for management decision support.
- A model component which contains the decision model representing expert knowledge encapsulated in a knowledge-base.
- A User Interface (UI) component as the DSS front-end. The UI is used by decision maker(s) to access and interact with the DSS, and as such is the most important component as it heavily influences how the DSS is going to be perceived and used.

6.3.3 Architecture of the decision support system

The DSS was developed on an *n*-tier client-server architecture in which the DSS presentation, processing and data management are logically separated into distinct processes (*Fig 6.6*). The logical architecture consists of: (*i*) a data server containing the DSS database(*s*) and the DBMS; (*ii*) an application server containing PHP scripts and HTML, CSS and JavaScript files representing the DSS back-end; (*iii*) a Web server that receives HTTP requests from the client and renders the HTML results back to the client; *and* (*iv*) a client representing the DSS front-end through which decision maker(s) have access to the DSS. The client interacts with the application layer by sending HTTP requests via the web server, which either processes the requests or parses the request to the database(s) or knowledge-base(s). The application server assembles the result(s) of the request and sends it to the web server, which renders the results as HTML output. The flow of information between the different DSS components is illustrated in *Figure 6.7*.

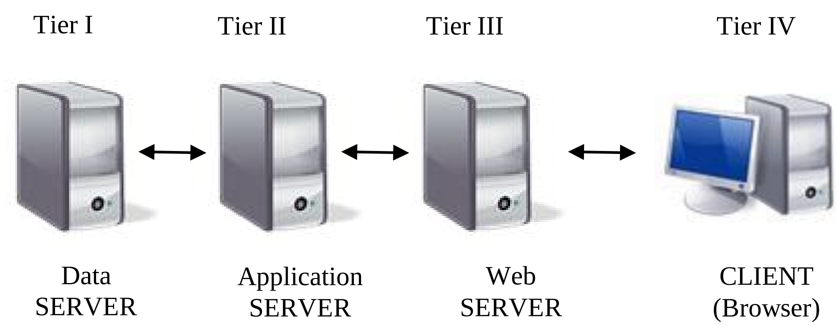

Figure 6.6 – n-Tier client-server architecture

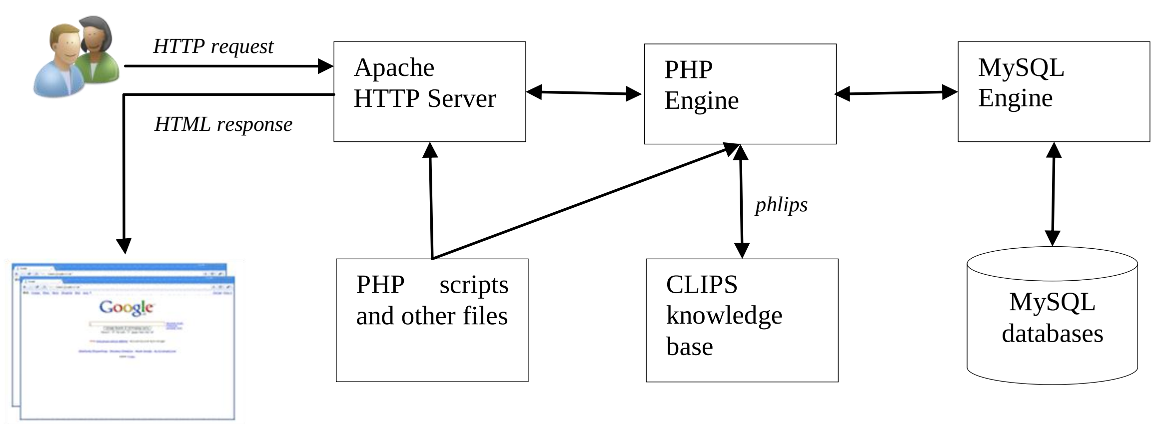

Figure 6.7 – The relationships between the different architectural components of the decision support system

6.3.4 Implementing the decision support system

The DSS was developed using the LAMP open source software[45] development stack. LAMP stands for Linux operating system, Apache HTTP[46] Server (Apache), MySQL

[45] Open source software is computer software which the source code is freely available in the public domain for use, redistribution and modification.

[46] HTTP (Hypertext Transfer Protocol) is an application protocol which defines how files on the World Wide Web are transferred. HTTP is the framework for how browsers will display and use file formats. When you

database server and PHP server-side scripting language, and is an open source software development stack that is a popular alternative to commercial, proprietary and platform specific development environments for developing powerful and dynamic Web database applications[47]. Linux is a generic name for Unix-type operating systems. Linux is known for its security and stability, making it an ideal choice for developing powerful and secure Web applications. The openSUSE Linux distribution was used for developing the DSS. Apache is a popular cross platform, free, secure, efficient and extensible server that provides HTTP services in accordance with the current HTTP standards (Apache 2009). Although primarily used for 'serving' Web pages, Apache supports many other features via 'modules' to extend its core functionality, ranging from server-side programming support (*e.g*. with PHP) to authentication (*e.g*. for MySQL authentication).

Web database applications have two constituent parts: (*i*) a database representing the memory; *and* (*ii*) an application that performs the tasks. In LAMP applications, MySQL provides the database functionality, and PHP provides the application functionality. MySQL and PHP are frequently used together because they are both Web orientated and have built in features for communicating with each other (Valade 2006). Apache is used for communicating between PHP and MySQL. The DSS was developed on an open source platform to allow free viewing, modification and distribution of the DSS source code, which should provide flexibility in integrating other components or with developing and additional components to the DSS.

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enter in a URL with HTTP at the beginning, you are requesting a web page which can contain other elements (such as pictures) and links to other resources.

[47] An application is a program designed for use by end user(*s*). If the end-user interacts with the application via a Web browser, the application is a Web based application. If the Web application requires the long-term storage of information using a database, it is a Web database application (Valade 2006).

6.4 DEVELOPMENT OF THE DATABASE COMPONENT

The core of the DSS is the database component which contains and manages the DSS database. The DSS database was designed as a Relational Database[48] (RDB) model, following the principles of RDB design, which involve: (*i*) identifying entities which represent facts to be stored; (*ii*) identifying the relationships between entities; (*iii*) identifying the attributes of each entity and their corresponding data type and size; (*iv*) assigning keys as unique identifiers of records in the database; *and* (*v*) normalising the RDB model to optimise its performance. The DSS database was then developed based on the optimised RDB model using the international standard database language, the Structured Query Language (SQL) embedded in MySQL Relational Database Management System (RDBMS) database server. The MySQL InnoDB storage engine was used for the DSS database, which is fully ACID[49] compliant. An overview of the design and development of the database are provided below. The detailed design, implementation and validation of the developed database are provided in Appendix IV.

6.4.1 Database entities

RDBs store data in relational tables called *entities*, which are searchable with a RDBMS. The properties of each entity in the RDB are known as its *attributes*, which are represented by the columns of the table. Data is stored in rows in RDB tables, with each row

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[48] Relational Databases (RDBs) are based on the relational model, which is a theoretical model of database systems that provide a means of representing data, the relationships between data items, and the way(s) in which the data may be used (Eaglestone 1991). One of the advantages of a relational database is that duplication of entries is reduced or even eliminated, allowing for the efficient management of large databases.
[49] In database design ACID stands for Atomicity, Consistency, Isolation and Durability which are a set of properties that ensure the reliability of database transactions. A transaction is a sequence of operations performed as a single logical unit, which must exhibit the four ACID properties.

representing an independent record. The data for the DSS was stored as relational tables, with each row representing an independent record, and each column representing its corresponding attribute. Each record has a unique identifier, data type and data size. The entities for the DSS were identified from publicly available DSS and *Decision Support Tools* (DSTs) for contaminated land management, published guidance and technical reports. The entities used in the database are presented in *Table 6.4*.

Table 6.4 – Entities in the database model

ENTITY	NOTATION	DESCRIPTION
Site	SITE	For storing site details
Preliminary qualitative assessment	PRELIM_QRA	For storing site details for preliminary risk assessment
Site samples	SAMPLE	For storing details of collected site samples
Environment Agency (EA) Soil Guideline Values (SGVs)	GAC_EA	Contains EA-based generic assessment criteria for stored contaminant in the database.
Generic quantitative risk assessment using EA SGV	GQRA_EA	For storing results of chemical analysis of measured site sample concentrations to be used for comparing with generic assessment criteria for risk assessment
Land use types	LAND_USE	Contains land use types. These are used for defining EA-based SGVs for different land-use scenarios.
Dutch Intervention Values (DIV) Generic Assessment Criteria (GAC)	GAC_DIV	Contains DIV-based generic assessment criteria for stored contaminant in the database.
Generic quantitative risk assessment using DIV	GQRA_DIV	For storing results of chemical analysis of measured site sample concentrations to be used for comparing with generic assessment criteria for risk assessment
Contaminated media	CONTAMINATED_MEDIA	Contains the type contaminated media to be assessed using DIVs.
Contaminants	CONTAMINANT	Contains details of all contaminants to be

		used for decision support, including indicator compounds and hydrocarbon fractions.
Contaminant type	CONTAMINANT_TYPE	Contains the different contaminant types.

The database includes both internal and external data. Internal data are stored in the database, and external data are supplied as input for the DSS. The internal data are stored in GAC_EA, GAC_DIV, LAND_USE_TYES, CONTAMINANTED_MEDIA and CONTAMINANT tables. The GAC_EA and GAC_DIV tables store the generic assessment criteria for Generic Quantitative Risk Assessment (GQRA). The GAC values that are used for GQRA in the DSS are based on published values from different sources: (*i*) the EA SGVs; (*ii*) the Land Quality Management/Chartered Institute of Environment Health (LQM/CIEH) GAC values; *and* the (*iii*) Dutch Intervention Values for the remediation of soil/sediment and groundwater (DIV 2000). The EA SGV and the LQM/CIEH values are stored in the GAC_EA table, and the DIVs are stored in the GAC_DIV table. The derivation of all the GAC values used in the database is provided in Appendix I. The contaminants stored in the database are based on the indicator compounds and hydrocarbon fractions in the UK framework for evaluating human health risks from petroleum hydrocarbon contamination.

6.4.2 Relationships between the entities

The relationships between the entities in the DSS are illustrated in the *Entit*y Relationship Diagram (ERD) in *Figure 6.9*. ERDs are conceptual schema of RDBs. The *Crow's Foot Notation* was used in the ERD to represent entities, relationships between the entities and their cardinality (Chen 1976). Entities are represented by boxes, and the relationships are represented by the lines between entities. The cardinalities represent the minimum and maximum number of entities in a relationship, and cardinalities are represented by three symbols: (*i*) a ring for representing ‘zero’; (*ii*) a vertical bar for representing ‘one’; *and* (*iii*) a crow’s foot representing ‘many’ or ‘more’. The symbols are used in pairs to represent four distinct types of relationships between entities (*Fig 6.8*). The cardinalities of all the relationships are represented in the ERD. The many-to-many relationships were solved using *associative entities* which store additional data that does not fit into the attribute list of either entity in the many-to-many relationship.

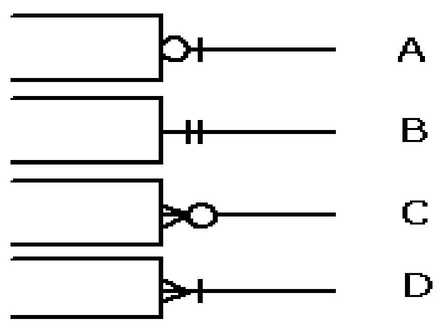

Figure 6.8 – The different cardinalities in Crow’s Foot Notation: A = zero-or-one; B = one-and-only-one, C = zero-or-many; and D = one-or-many

6.4.3 Attributes and their data types

An attribute is a unit of fact that describes the properties of an entity. Attributes are represented by the columns in a table. Each row of an entity has a value for each of its attributes (which could be null) with each row having the same data type and size for the same attribute. The attributes of each of the entities in the database are presented in *Table*

6.5. A description of all the attributes for all the entities identified for is given in Appendix IV. These are based on the input data that are required for each stage of the contaminated land management decision-making process. These were identified from publicly available contaminated land DSSs and DSTs and published guidance and technical reports. Each record has a data type specified by its attributes. Data types determine which type of information will be stored for each attribute, its value, and its size. Different numeric, date/time and string data types have been used for the attributes, with varying sizes. All the data types used in the DSS and their respective values are provided in *Appendix VI*.

Table 6.5 – Entities and their corresponding attributes

ENTITY	ATTRIBUTES
SITE	id, site_name, site_loc, site_description, authors
PRELIM_QRA	site_id, c_type, c_luse, l_euse, nnluse, snluse, wluse, enluse, perm, vuln, gwater, swater, abs_lic
SAMPLE	id, site_id, sample_name, sample_media, sample_description
GAC_EA	id, contaminant, land_use, gac_value
GQRA_EA	id, sample_name, contaminant, gac_value, ms_conc0, ... , ms_conc34
LAND_USE	id, land_use_type
GAC_DIV	id, contaminant, media, div_value
GQRA_DIV	id, sample_name, contaminant, div_value, ms_conc0, ... , ms_conc34
CONTAMINATED_MEDIA	id, media
CONTAMINANT	id, name, type_id
CONTAMINANT_TYPE	id, type

6.4.4 Assigning keys as identifiers

In RDBs, unique identifiers known as *keys* are used for enforcing database integrity. Each table in the DSS database has a Primary Key (PK), which uniquely identifies each record in the database. Foreign Keys (FKs) have been used to connect tables together. An FK is an attribute of a table that is used as the PK of another entity. The FK and the referenced PK must have the same data types, and ideally size. FKs are used to ensure referential integrity across the database is maintained by ensuring all data is cross-referenced from within the database is also described within the database. This reduces data duplication and redundancy (Eaglestone 1991). All the FKs in the database and their referenced PK were indexed to allow for quick checking of data integrity, optimising database performance.

6.4.5 Normalisation of the database model

Normalisation is a process of optimising an RDB model by refining and organising data to ensure all data dependencies are logical. Normalising database reduces data redundancy and operational anomalies, and improves the overall efficiency and performance of the database. Normalisation modifies the RDB model using a series of progressive restrictions, called *Normal Forms* (NFs), each of which progressively excludes certain undesirable properties from the database design (Eaglestone 1991). Many NFs have been defined, of which there are six well established NFs in database theory (Eaglestone 1991), starting the lowest, the first normal form (1NF) to the highest form, the fifth normal form (5NF).

In most practical applications, RDB models are only normalised to 3NF, as there is a trade-off between complete normalisation and database performance. The more progressively normalised an RDB model is, the more tables it will contain, which results in more SQL operations, potentially leading to decrease in operational performance. To that effect, the DSS RDB model has been normalised to 3NF, with all the tables in the database satisfying

1NF, 2NF and 3NF requirements. 1NF requires all the tables to have a PK, and contain no repeating values within any column of all the tables. 2NF requires all *non-key* columns to be dependent on the entire PK of the table, and for composite PKs, all *non-key* columns must depend on the whole not part of the PK. All the tables are in 2NF because all the *non-key* attributes of each of the tables in the RDB model are fully dependent on the respective table's PK. 3NF requires that all columns in a table directly depend on the PK of that table and not on other attributes. The NFs are cumulative, with each higher form depending on meeting the requirements of its lower form, therefore all 3NF tables satisfy 1NF and 2NF requirement. The ERD of the normalised database model showing entities, their attributes, relationships and keys is provided in *Figure 6.9*. The normalised database model was implemented in a MySQL RDBMS. A detailed design, normalisation, implementation and validation of the DSS database is provided in *Appendix IV*.

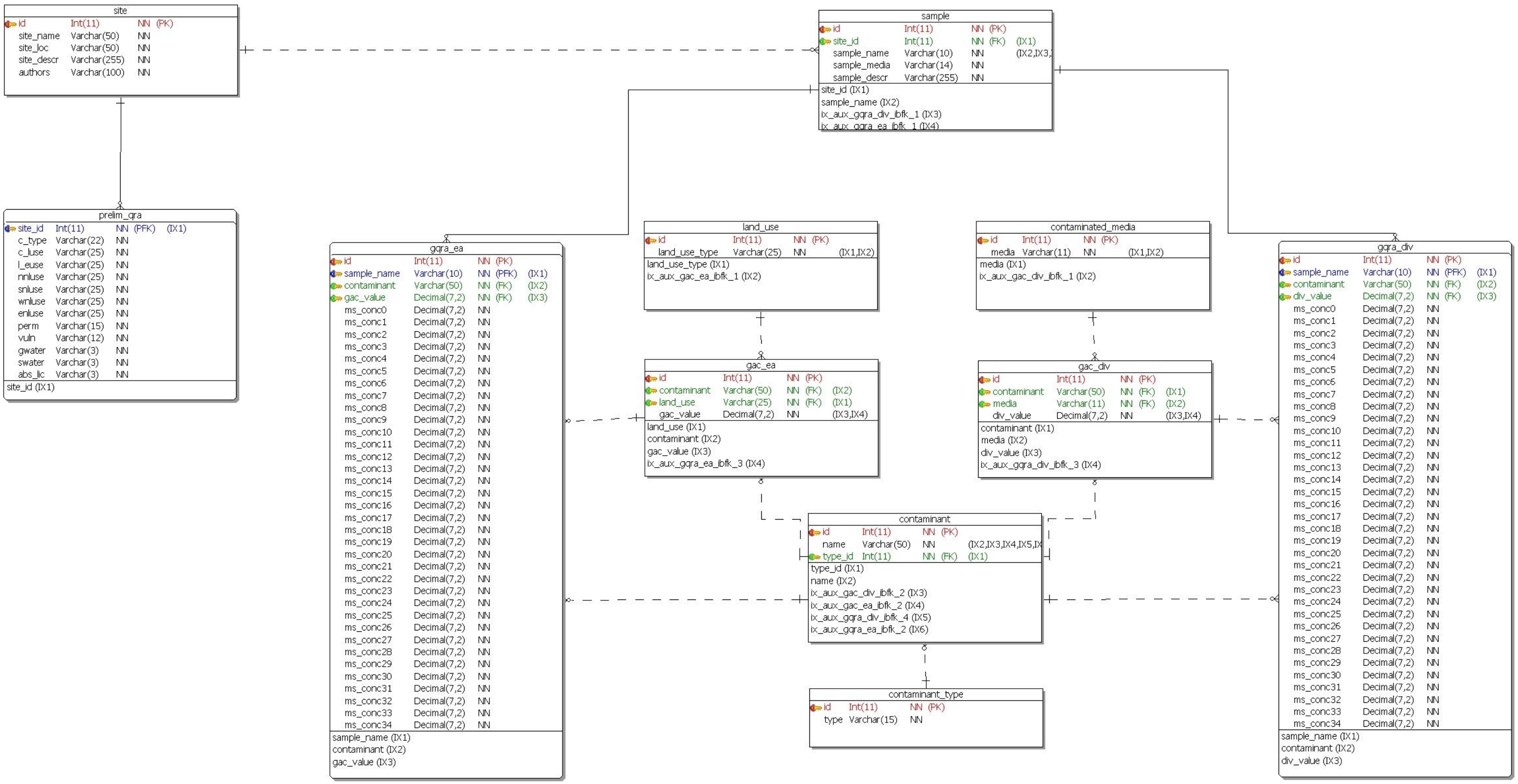

Figure 6.9 – The normalised entity relationship diagram for the relational database model showing entities with their corresponding attributes, data types, and keys

6.5 DEVELOPMENT OF THE MODEL COMPONENT

A decision model encapsulating judgement on the relative ranking of remediation technologies was developed for the DSS. The decision model was developed using the Analytical Hierarchy Process (AHP), a structured Multi Criteria Decision Analysis (MCDA) method that is used for systematically comparing decision outcomes. The AHP was used to weigh and rank remediation technologies based on the sustainability criteria. The result of the AHP decision model was encapsulated in a knowledge-base. The knowledge-base was developed using the CLIPS expert system shell[50]. CLIPS stands for C Language Integrated Production System, and was originally developed by NASA and is now freely available in the public domain. CLIPS is a complete environment for the development and delivery tool for knowledge-base systems (CLIPS 2008). The knowledge-base was integrated with the rest of the DSS using the PHP CLIPS extension, the PHLIPS extension, which allows the deployment of knowledge-bases in PHP by providing access key functions in the CLIPS library, such as loading CLIPS program file(*s*), executing them and retrieving the results (PHLIPS 2005).

6.5.1 Development of the decision model

The AHP process broadly consists of four key stages: (*i*) problem formulation; (*ii*) weights valuation; (*iii*) weights aggregation; *and* (*iv*) sensitivity analysis. In developing the decision model, the decision goal, alternatives and criteria were first identified (*Table 6.6*). The goal of the decision model is the selection of the most sustainable technology, given site

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[50] A shell is a software that provides a suitable framework within which knowledge can be held and manipulated: the shell itself is empty of knowledge (Finlay 1990).

parameters, for the remediation of contaminated land. Both the sustainability criteria and the remediation technologies (alternatives) used in the DSS were derived from the literature, technical reports and expert judgement. All the alternatives identified are established remediation technologies that are suitable for cleaning petroleum hydrocarbon contamination (Friend 1996, FRTR 2010, CLU-IN *c*2010, EUGRIS *c*2010). The sustainability criteria are based on the indictors for sustainable remediation identified by the Sustainable Remediation Forum UK (SURF-UK). Descriptions of all the remediation technologies and the sustainability criteria are provided in Appendices II and III.

Table 6.6 – The sustainability criteria, sub-criteria and alternatives used in the decision support system

CRITERIA	SUB-CRITERIA	ALTERNATIVES
Economic	Direct costs (EC1)	Bioventing (A1)
	Indirect costs (EC2)	Enhanced bioremediation (A2)
	Time span (EC3)	Monitored natural attenuation (A3)
Environmental	Impacts on resources (EN 1)	Phytoremediation (A4)
	Impacts on ecological systems (EN2)	Air sparging (A5)
	Intrusiveness (EN3)	Soil vapour extraction (A6)
	Resource use and waste by-products (EN4)	Thermal treatment (A7)
Social	Impacts on human health (S1)	Soil washing (A8)
	Impacts on neighbouring land (S2)	Incineration (A9)
	Uncertainty, evidence and policy(S3)	Thermal desorption (A10)
		Excavation and disposal (A11)

The decision goal, criteria, sub-criteria and alternatives were decomposed into a four level hierarchical structure (*Fig 6.10*). The hierarchy provides an overall view of the relationships within the different elements of the decision problem and allows for the comparison elements of the same order of magnitude with respect of the overall goal (Saaty 1987). A four level hierarchy was developed because it has been observed that criteria with a large number of sub-criteria tend to receive more weight than when they are less detailed, it is recommended that for hierarchies with large numbers of elements, the elements should be arranged in clusters so they do not differ in extreme ways (Ishizaka and Ashraf 2009).Both qualitative and quantitative information were used for pairwise comparisons at each hierarchical level. The pairwise comparisons were carried out using the Saaty fundamental 9-point scale of absolute numbers which is used to assign numerical values to both quantitative and qualitative judgements by asking questions like 'with respect to criterion ***x***, how much more important or dominant is alternative *a* to *b*?' (*Table 6.7*).

Table 6.7 – Saaty's fundamental 9-point scale for pairwise comparisons

Intensity of importance	DESCRIPTION
1	Criterion i and j are of equal importance
3	Criterion i is moderately more important than criterion j
5	Criterion i is strongly more important than criterion j
7	Criterion i is very strongly more important than criterion j
9	Criterion i is extreme more important than criterion j
2, 4, 6, 8	For compromise between above values

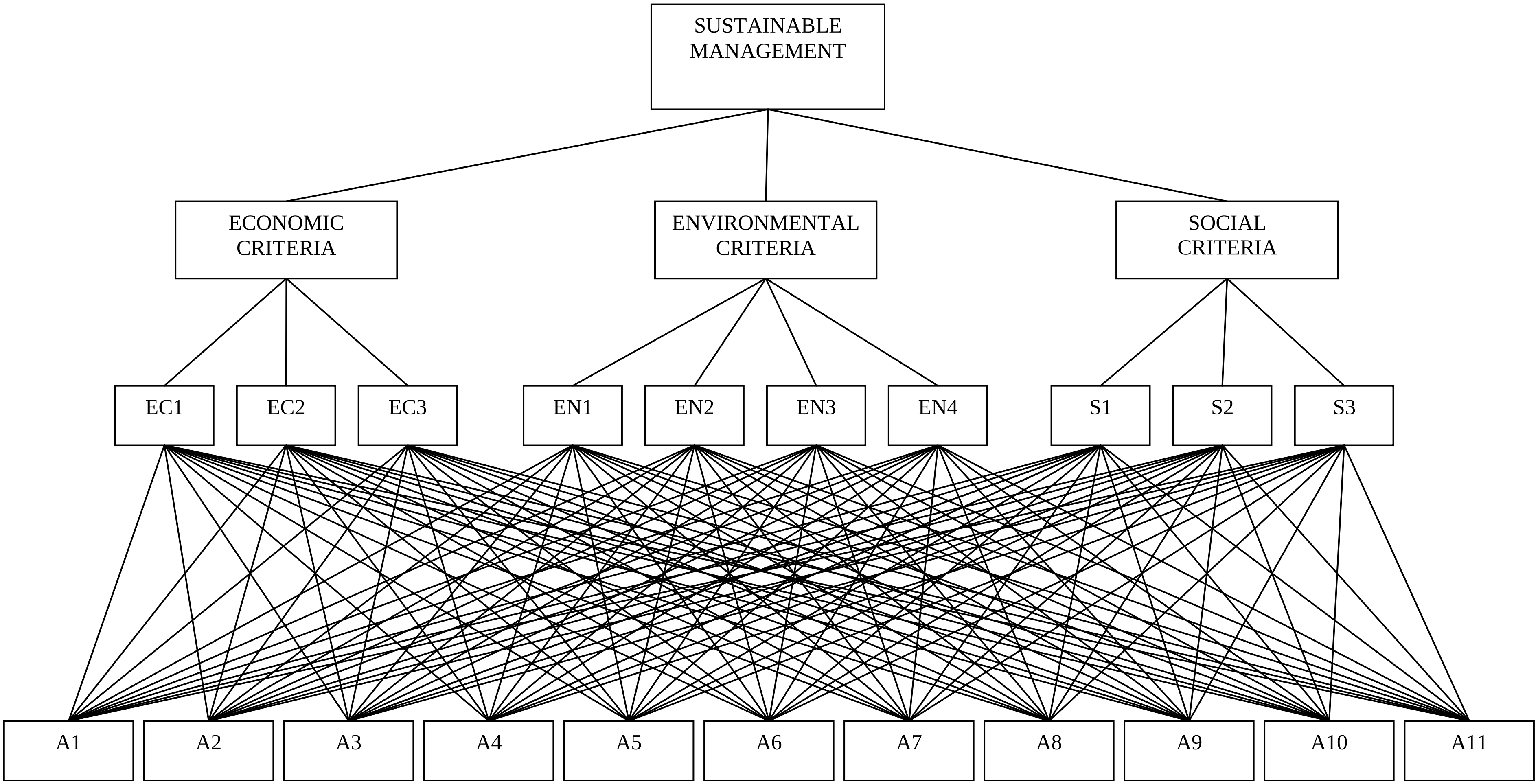

Figure 6.10 – The decision hierarchy for the selection of the most sustainable remediation technology

The elements of each hierarchical level were prioritised based on their relative importance to every other element in their hierarchy, with respect to a common parent element, *i.e*. criteria were compared with respect to goal, sub-criteria to each of their parent criteria, and alternatives with respect to each sub-criterion. The weights were assigned to criteria, according to their relative importance based on expert opinion and literature. The results of the pairwise comparisons were recorded in positive consistency matrices. The overall priorities at each level of the hierarchy = 1.0. The decision model contains a total of 14 pairwise comparison matrices, consisting of a total of 556 pairwise comparisons. After the comparison matrices were completed, priorities were derived using the original AHP method, the Eigen value method, by normalising each column of each matrix, to derive the normalised principal Eigen (priority) vector. After that, the consistency of each of the comparison matrices was calculated. The local priorities across all the criteria were aggregated and normalised to derive their overall priorities. In the last stage of the AHP process, sensitivity analysis was carried out, which involved slightly modifying the weights of the criteria to observe the impact on the priority weights.

Table 6.8 – Pairwise comparisons of sustainability criteria with respect to overall goal

	Eco.	Env.	Soc.
Eco.	1.0	$\frac{1}{3}$	$\frac{1}{5}$
Env.	3.0	1.0	½
Soc.	5.0	2.0	1.0

The derivation of priorities is demonstrated with the first comparison matrix, the pairwise comparisons of sustainability criteria with respect to the goal (*Table 6.8*). All the other comparison matrices are provided in Appendix V. A value of 1.0 is recorded in the comparison matrix when an alternative is compared to itself, such that: $a_{ii} = 1$. The number of judgments J that were made in the comparison matrix was $J = \frac{n(n-1)}{2}$ = 3, where n is the size of the matrix (Saaty 1990). This is because only the upper triangular matrix needs to be filled in – the bottom triangular matrix is derived as the reciprocal value of the upper matrix such that for all $a_{ij} = k$, the corresponding diagonal reciprocal value is $a_{ji} = \frac{1}{k}$.

Table 6.9 – The relative weights of the pairwise comparisons

	Eco.	Env.	Soc.
Eco.	1.0	$\frac{1}{3}$	$\frac{1}{5}$
Env.	3.0	1.0	½
Soc.	5.0	2.0	1.0
∑ COL	9	$\frac{10}{3}$	$\frac{17}{10}$

Table 6.10 – The normalised relative weights of the pairwise comparisons

	Eco.	Env.	Soc.
Eco.	$\frac{1}{9}$	$\frac{1}{10}$	$\frac{2}{17}$
Env.	$\frac{3}{9}$	$\frac{3}{10}$	$\frac{5}{17}$
Soc.	$\frac{5}{9}$	$\frac{3}{5}$	$\frac{10}{17}$
Norm. COL	1.0	1.0	1.0

All the elements of the completed matrix are positive and reciprocal such that $a_{ij} > 0$. The normalised relative weights were calculated by adding the values of each column of the reciprocal matrix (*Table 6.9*), and then dividing each value of the column by the sum, which = 1.0 (*Table 6.10*). The priority vector, the principal Eigen vector, was computed by averaging each row of the comparison matrix using $A \times w = \lambda_{max} \times w$, where A is the comparison matrix; w is the normalised principal Eigen vector; and λ_{max} the priority value of A (Saaty 1987). The normalised priority vector w was obtained by averaging across the rows (*Table 6.11*). The sum of all the elements of the priority vector = 1.0. The priority vector represents the relative weights of the criteria with respect to the overall goal, which for the comparison matrix is 58%, 31% and 11% for the social, environmental and economic criteria respectively. In most cases, the sustainability criteria are given equal weights, however in this case the sustainability criteria have different weights representing the influence of each criterion to the decision problem, which rates the protection of human health (a social sub-criterion) above all other criteria in the decision hierarchy.

Table 6.11 – The normalised principal Eigen (priority) vector

	Eco.	Env.	Soc.		∑ ROW			Eigen vector
Eco.	$\frac{1}{9}$	$\frac{1}{10}$	$\frac{2}{17}$		0.329			0.10959
Env.	$\frac{3}{9}$	$\frac{3}{10}$	$\frac{5}{17}$	=	0.927	*	$\frac{1}{3}$	0.30915
Soc.	$\frac{5}{9}$	$\frac{3}{5}$	$\frac{10}{17}$		1.744			0.58126
∑ COL	1.0	1.0	1.0					∑ = 1.0

Finally, the consistency of the comparison matrix was calculated. Although the AHP allows for inconsistency in decision-making, the AHP provides a method of calculating the decision maker(s) inconsistency, the Consistency Index (*CI*) which is used to determine the degree of consistency in a comparison matrix. A threshold value of ≤ 0.10 is deemed acceptable. A larger CI value will disrupt consistent measurement, and lower CI value would make an insignificant change in measurement (Saaty 2004, Saaty 1990). Other methods have been developed for deriving such priorities in an effort to reduce rank reversal. The most common of which is the geometric mean (*also* logarithmic least squares) method (Ishizaka 2004, Ishizaka and Labib 2009). It has been mathematically demonstrated that the Eigen vector solution is the best approach (Saaty 1990).

A comparison matrix is consistent if for all i, j, k the ranking is transitive, such that: $a_{ij} * a_{jk} = a_{ik}$. In consistent reciprocal matrices, the principal Eigen (priority) value $\lambda_{\max}$ should be equal to the size of the comparison matrix *n*, such that: $\lambda_{\max} = n$. The principal Eigen value is calculated by multiplying the Eigen vector with the sum of the criteria weights of each of its reciprocal matrix and then adding all the products:

$$\lambda_{\max} = 9(0.10959) + \frac{10}{3}(0.30915) + \frac{17}{10}(0.58126) = 3.004952$$

The CI is calculated as $CI = \frac{\lambda_{\max} - n}{n-1}$, where $\lambda_{\max}$ is the principal Eigen value and *n* is the dimension of the comparison matrix. The CI of the comparison matrix was calculated:

$$CI = \frac{3.004952 - 3}{3 - 1} = 0.002476$$

The Consistency Ratio (CR), which is the ratio between CI and the *Ratio Index* (RI), was then calculated using: $CR = \frac{CI}{RI}$ (*Table 6.12*). The RI is the average CI of 500 randomly filled matrices (Saaty 1977). Other RI values have been calculated by other researchers, and alternative methods exist for measuring consistency (Ishizaka and Labib 2009).

Table 6.12 – Random index values (Saaty 1977)

N	1	2	3	4	5	6	7	8	9	10
RI	0.00	0.00	0.58	0.90	1.12	1.24	1.32	1.41	1.45	1.49

$$CR = \frac{0.002476}{0.58} = 0.00427$$

The $CR = 0.00427$ < 0.1 %, therefore the comparison matrix is considered consistent. This result was validated using the Expert Choice 11.5™ (EC) AHP software (EC 2009). The remaining 13 comparison matrices were calculated the same way and validated using EC, with number of pairwise comparisons and the level of difficulty increasing as the size of the matrix increased. The comparison matrices of their CI are presented in Appendix V. After all the comparison matrices have been completed and their consistencies checked with the EC software, the overall priority of the alternatives was derived by aggregating the local priorities across all criteria using (Ishizaka and Labib 2009):

$$p_i = \sum_j w_j * l_{ij}$$

Where p_i is the overall priority of alternative i,

l_{ij} the local priority, *and*

w_j weight of the criterion j.

Two approaches are used for deriving global priorities: (*i*) the ideal mode; *and* (*ii*) the distributive mode, which do not necessarily provide the same ranking. The ideal mode normalises by dividing the score of each alternative only by the score of the best alternative under each criterion. This prevents rank reversal and is suited for decision models that might change with addition and/or deletion of criteria or alternatives. Rank reversal occurs when judgements are altered when alternatives are added or deleted, even when the additions are irrelevant and deletion does not result in loss of information. Advocates of utility theory argue that adding alternatives, even irrelevant ones, should not cause rank reversal (Saaty 1990). AHP proponents however consider rank reversal as an asset as it mirrors normal human behaviour. Moreover, rank reversal phenomenon is not unique to AHP but to all additive models (Ishizaka and Labib 2009).

Figure 6.11 – Overall synthesis with respect to goal in ideal mode

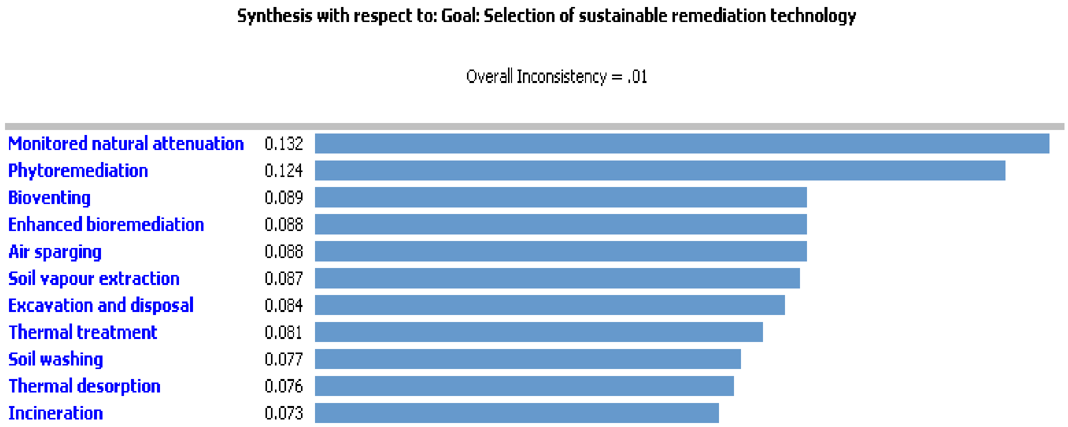

Figure 6.12 – Overall synthesis with respect to goal in distributive mode

The distributive mode is suitable when the priorities are known. However, rank reversal may occur when alternatives are added and/or deleted, even if they are a copy of an existing alternative. Because the decision model might change with the addition and/or deletion of new criteria or alternatives, the ideal mode was used in the EC software so as to prevent rank reversal. The global priorities of the alternatives in the ideal and the distributive modes are presented in *Figures 6.12* and *6.13* respectively. As can be seen from the diagrams, although the priorities are different, the ranking is almost identical, with the exception of thermal treatment and excavation and disposal ranks reversing.

6.5.2 Encapsulation of the decision model in a knowledge-base

The developed AHP decision model was encapsulated in a knowledge-base, which was developed using the CLIPS expert system shell. In expert systems, knowledge is regarded as any construct and fact about the problem the system is trying to solve (Finlay 1990). CLIPS is a forward chaining rule-based tool that provides the basic elements of knowledge-base component: (*i*) a fact-list as the global memory for data; (*ii*) a rule-base which contains all the rules; *and* (*iii*) an inference engine which controls overall execution of the rules (*Fig 6.11*). CLIPS programs can be integrated with other applications/programs/programming languages and can both be embedded into external programs or for calling other programs from CLIPS. This allows for the easy integration of CLIPS programs with existing programs, and is especially useful in cases where the CLIPS program is a small or a larger task/system, or needs to share data with other functions (CLIPS 2007*b*).

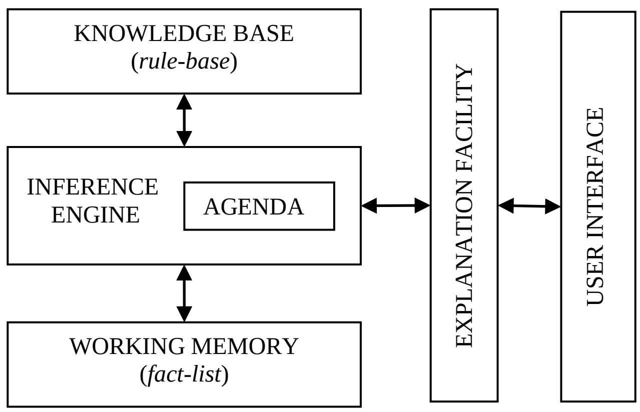

Figure 6.13 – Structure of the decision support system knowledge-base

Facts are the fundamental unit of data, with each fact representing a piece of information (CLIPS 2007*b*). Rules are composed of an IF portion and a THEN portion. The IF portion are conditions which must be satisfied for the rule to be applicable. The conditions are satisfied based on the existence or non-existence of specified facts in the fact-list. The THEN portion of a rule is the set of actions to be executed when a rule is applicable. The process of matching facts to rules is called pattern matching, and is carried out by the inference engine, which infers which rules should be executed and when (CLIPS 2007*a*).

The facts for the knowledge-base are based on the remediation technologies and sustainability criteria from the AHP decision-model derived from published technical reports and guidelines. The rules rank remediation technologies based on selected criteria. The technologies are ranked according to assigned certainty values, which were derived from the relative ranking of the technologies from the AHP decision model (*Fig 6.12*). The knowledge-base contains a total of 2407 rules, for all possible selections of the sustainability criteria. This was confirmed by the additive combination equation: $C(n,r) = \frac{P(n,r))}{r!} = \frac{n!}{r!(n-r)!}$ where: C is the combination equation, n the number of alternatives *and* r the number of combinations. *E.g*. the equation $C(11,3) = \frac{11!}{3!(11-3)!} = 165$ was used for determining all possible combinations of 3 alternatives from the total of 11. All possible combinations of 1, 2, 3, 4, 5, 6, 7, 8, 9, 10 and 11 alternatives was calculated and summed. The decision-makers access the knowledge-base using the UI. The explanation facility provides the result of the ranking of the remediation technologies. The certainty values are cumulative, with the value increasing with increasing number of selections, *i.e*. the more criteria are selected the higher the certainty value of each technology will be.

Before facts can be created, the type of value that field(s) can contain is explicitly declared, with groups of facts that share common information described using the *deftemplate* construct (CLIPS 2007*b*). Constructs form the core of CLIPS programs, and are used for adding facts and rules into the knowledge-base (Giarratano and Riley 1989). It is convenient to automatically assert a set of facts instead of typing in typing in the same assertions from the top level, particularly for facts that are known to be true, such as with the developed knowledge-base. Groups of facts that represent initial knowledge are defined using the *deffacts* construct. In order to accomplish useful work, the CLIPS program must have rules as well as facts, which can be typed directly into clips or loaded from a file of rules (Giarratano and Riley 1989). A rule is composed of an *antecedent* and a *consequent*. The antecedent of a rule is the IF portion or the left-hand side (LHS) of the rule, and the consequent of a rule is the THEN portion or the right-hand side (RHS) of the rule. The antecedent of a rule is a set of conditions (or conditional elements) which must be satisfied based on existence or non-existence of specified facts in the fact-list for the rule to be applicable. The consequent of a rule is the set of actions to be executed when the rule is applicable. The inference engine is used in CLIPS to automatically match patterns against the current state of the fact-list and determines which rules are applicable (CLIPS 2007*b*). The *defrule* construct is used for defining rules.

As an example, the ranking of the remediation technologies based on multiple criteria (direct cost, impacts on resources and impacts on human health) is expressed by the *pseudocode* below. If all the criteria are selected, the ranking of the technologies and the corresponding certainty values will correspond to the TOTAL column of *Table 6.13*.

```
IF sub-criteria direct-cost
AND sub-criteria impacts-on-other-resources
THEN technology-is monitored-natural-attenuation with certainty 17
   AND technology-is phytoremediation with certainty 16
   AND technology-is excavation-and-disposal with certainty 8
   AND technology-is soil-vapour-extraction with certainty 11
   AND technology-is bioventing with certainty 11
   AND technology-is enhanced-bioremediation with certainty 11
   AND technology-is air-sparging with certainty 11
```

```
AND technology-is thermal-treatment with certainty 6
AND technology-is soil-washing with certainty 6
AND technology-is thermal-desorption with certainty 4
AND technology-is incineration with certainty 2
```

The rule can be converted into a rule by defining a *deftemplate* for the types of facts referred by the rule, where the sub-criteria and remediation technologies can be represented by the *deftemplate* below. The `type` and `weight` fields contain the selected sub-criteria and the weights for the remediation technologies based on the selection respectively.

```
(deftemplate sub-criteria (slot type))
(deftemplate technology-is (slot weight))
```

The general format of a fact is:

```
(deffacts <deffacts name> [<optional comment>]
      <facts>*)
```

The general format of a rule is:

```
(defrule <rule name> [<optional comment>]
      [<declaration>]
      <conditional-element>*  ; LHS of the rule
      = >
      <actions>*              ; RHS of the rule
```

The PHP extension PHLIPS has been used to interface the knowledge-base with the DSS UI by calling and executing the CLIPS program using the PHP code. Once the sustainability criteria have been selected, PHP loads the CLIPS program and inserts the selection into the fact-list and fires the appropriate rule(s) and displaying the results.

Table 6.13 – Relative ranking of remediation technologies expressed as certainty values

	Direct costs	Indirect costs	Time span	Impacts on other resources	Impacts on ecological systems	Intrusiveness	Resource use and waste by-products	Impacts on human heath	Impacts on neighbouring land	Uncertainty, evidence and policy	TOTAL
Monitored natural attenuation	7	1	1	10	4	7	5	1	1	1	38
Phytoremediation	7	1	1	9	4	6	5	1	1	2	37
Bioventing	4	4	2	7	3	4	4	1	1	1	31
Enhanced bioremediation	4	4	4	7	3	3	3	1	1	1	31
Air sparging	3	5	2	8	3	5	2	1	1	1	31
Soil vapour extraction	5	3	2	6	3	5	2	1	1	1	29
Thermal treatment	2	6	3	4	3	3	1	1	1	1	25
Excavation and disposal	6	2	5	2	2	1	3	1	1	1	24
Soil washing	1	7	4	5	1	2	1	1	1	1	24
Thermal desorption	1	7	4	3	2	2	1	1	1	1	23
Incineration	1	7	4	1	2	2	1	1	1	1	21

6.6 DEVELOPMENT OF THE USER INTERFACE

After the development of the database and the model components, the UI of the DSS was designed and developed following the current international standards of UI design provided in ISO 9421, particularly parts 151 and 171 which cover guidance on World Wide Web (WWW) UIs and issues of general software accessibility respectively (ISO 2008*a*, ISO 2008*b*). A good UI design is essential because to most users' (the decision-makers), the UI is the DSS. A well designed UI can significantly increase the effectiveness of the support provided by the DSS. ISO 9241-151 provides principles of content design and presentation, user navigation and interaction of Web applications representing different levels of the design process (ISO 2008*a*). ISO 9241-171 provides guidance on designing accessible software (ISO 2008*b*).

Both parts of the ISO 9421 used in developing the DSS UI provide procedures for evaluating that the ISO recommendations have been followed, which was used for evaluating the UI design. A user friendly UI was designed for the DSS as a *Graphical User Interface* (GUI). This provides better accessibility of the DSS compared to a *Command Line Interface* (CLI). Both the database and the model components have CLIs, and the DSS GUI provides a common interface that integrates the DSS components. The design and development of the DSS UI was done following established principles of interface design; DSS interface design and *Human-Computer Interaction* (HCI). These are (Faulkner 1998):

- *Naturalness* – the GUI was designed in a way to reflect the decision maker(*s*) syntax and semantics, *i.e*. it is in the natural language of the tasks involved and is structured according to that task. The GUI interface does not require any human pre- or post-processing of the input parameters or the output of the DSS.

- *Consistency* – the GUI is consistent in its requirements for input and has consistent mechanisms for the decision maker using the DSS. All the DSS menus, messages, cancellation of commands and prompts are the same and are placed in the same position. The GUI provides validation of the inputs provided in the same format.
- *Relevance* – the GUI requires only the minimum user input and provides the minimum output necessary to support decision-making, and does not request for any redundant information. Most of the inputs required are database parameters, which prevents unnecessary errors. The DSS does not offer or request information that it can derive from previous inputs, or anything that it will not use; only information that is necessary for the completion of tasks is required.
- *Supportiveness* – the GUI provides adequate information to support users with operating the DSS. The GUI provides specific help to guide users with input requests, and provides adequate status feedback on completion or failure of tasks. The GUI content and navigational structure was designed in accordance with Web-navigation principles. Navigation between the pages is provided at the top of each page, where the user can see where they are in the decision-making process.
- *Flexibility* – the GUI accommodates differences in user requirements, preferences and level of performance and provides site-specific output headers; comparison of site-specific measured concentrations; comparison with different Generic Assessment Criteria (GAC); and flexibility in selecting remediation design parameters, remediation technologies, sustainability criteria and sub-criteria.

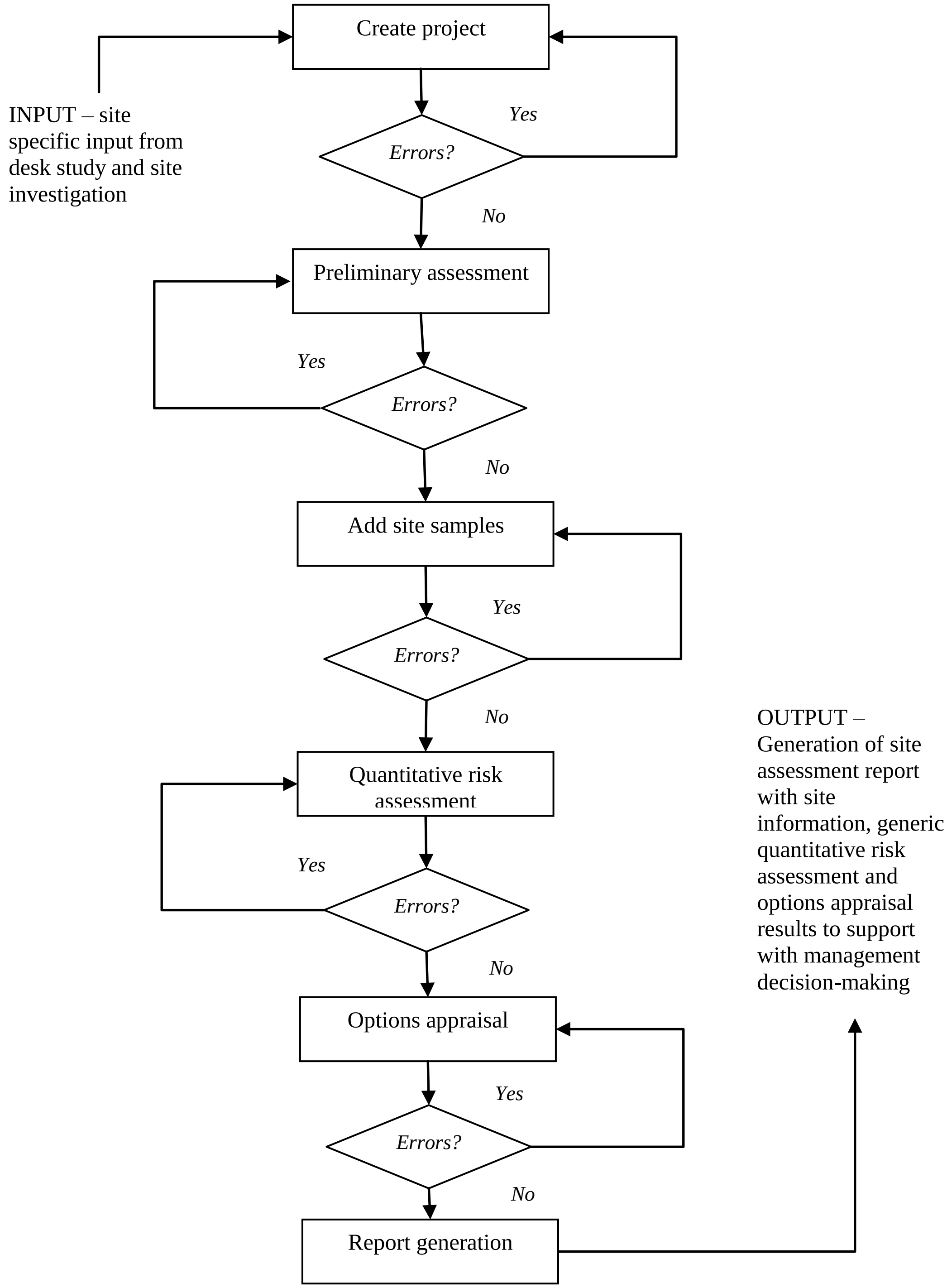

Figure 6.14 – The decision-making process of the decision support system

6.6.1 *Creating new projects*

The decision support process begins with creating a new project, where all inputs will be saved to the same project (*Fig 6.16*). The decision-making process of the DSS is illustrated in *Figure 6.15* above. The site details provided will be used for generating a site specific assessment report at the end of the management decision-making process. Each site assessment and/or management task is part of the project. As all the input fields in the form are supplied by the user, the form validates the inputs to check the required fields have been provided and all the fields are in the correct format required by the database. Two buttons are provided at the end of the form, one for resetting the form and the other for creating the project. A confirmation is provided if the project has been successfully created, otherwise a list of error messages is provided if errors are detected.

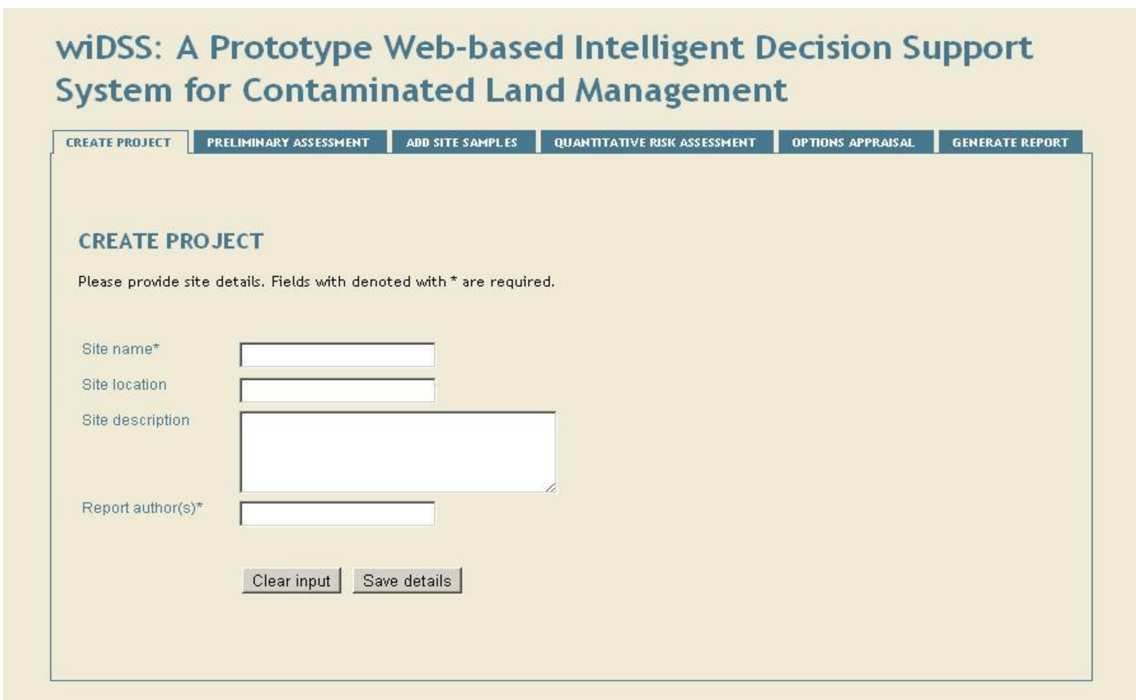

Figure 6.15 – Main page of the decision support system

6.6.2 Preliminary (qualitative) risk assessment

Preliminary qualitative risk assessment can be carried out after a project has been created.

The input form for this is provided in *Figure 6.17*. All of the input fields are parameters generated from the database, and are site-specific information of current land use, land end-use, neighbouring land uses and information about nearby water resources, abstraction license. Two buttons are provided at the end of the form, one for resetting the form and the other for adding preliminary assessment information to the project.

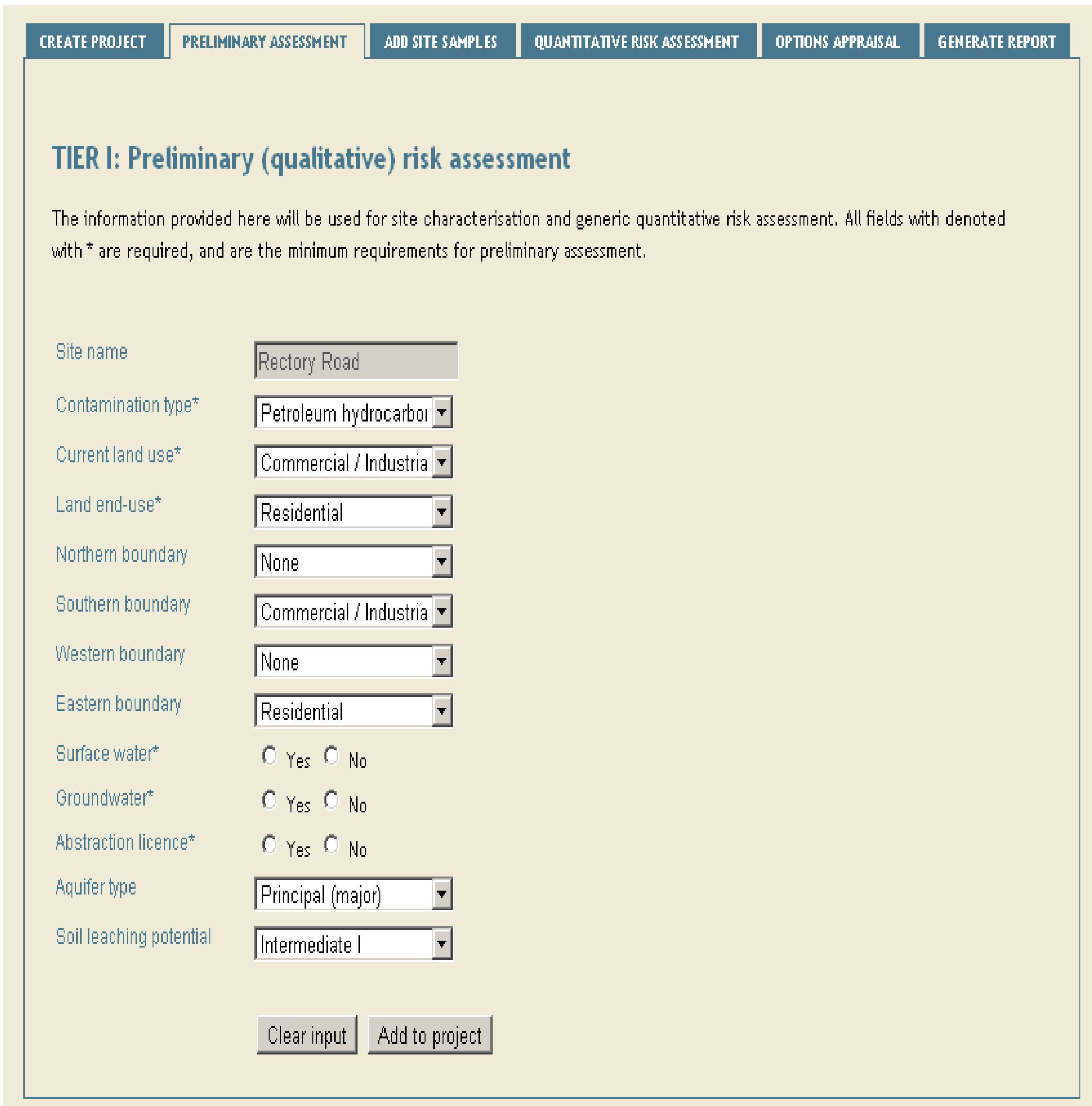

Figure 6.16 – Preliminary (qualitative) risk assessment of the decision support system

Land use information is used to determine which regulations might apply during the management life cycle, and to determine the appropriate GAC values for comparing measured site sample concentrations. For example, the planning and or building regulations might apply if site end-use is different from current use. Three different generic land use types have been adopted in the DSS, based on DEFRA and EA SGVs, which are based on generic assumptions of human behaviour and characteristics of contaminant exposure pathways for a range of different scenarios: (*i*) residential; (*ii*) allotments; and (*iii*) commercial/industrial land use. The GAC values are based on consideration of oral, dermal and inhalation routes of exposure. The toxicological effects are considered to be systematic and the combined assessment criteria for oral and dermal pathways are used. The GAC values for residential and allotment land uses are based on estimates for young children because they are generally more likely to have higher exposures to soil contaminants.

Neighbouring land use types are considered on the northern, southern, western and eastern boundaries to ensure sensitive land uses and receptors are protected. For example, even if land use change is to a less sensitive land use type, and more conservative GAC values may be selected by the DSS if a there is a more sensitive neighbouring land with strong pollutant pathways. The presence of groundwater, surface water and/or aquifer on site might also present potential receptors from the contamination source(s). The pathway between the contaminant source(s) and the receptors is determined by the soil leaching potential. The vulnerability of groundwater to contamination is determined by the soil leaching potential and the aquifer type. The EA has classified soil leaching potential (in England and Wales) into three soil vulnerability cases and six sub-classes (*Table 6.14*), reflecting the ability of contaminants to leach through the covering soils and pose a potential risk to groundwater at depth.

Table 6.14 – Soil vulnerability classification (EA 2009c)

	Soils with high leaching potential (H) – include soils with little ability to attenuate diffuse source pollutants and in which non-absorbed diffuse source pollutants and liquid discharges have the potential to move rapidly to underlying strata or to shallow groundwater. This includes three sub-classes:	
1	H1:	Soils that readily transmit liquid discharges because they are either shallow or susceptible to rapid by-pass flow directly to rock, gravel or groundwater
2	H2:	Deep, permeable, coarse textured soils which readily transmit a wide range of pollutants because of their rapid drainage and low attenuation potential
3	H3:	Coarse textured or moderately shallow soils which readily transmit non-absorbed pollutants and liquid discharges but which have some ability to attenuate adsorbed pollutants because of their large clay or organic matter contents.
	Soils with intermediate leaching potential (I) – soils which have a moderate ability to attenuate diffuse source pollutants or in which it is possible that some non-absorbed diffuse source pollutants and liquid discharges could penetrate the soil layer This include two sub-classes:	
4	I1:	Soils which can possibly transmit a wide range of pollutants
5	I2:	Soils which can possibly transmit weakly or non-adsorbed pollutants and liquid discharges but are unlikely to transmit adsorbed pollutants
6	Soils with low leaching potential (L) – soils in which pollutants are unlikely to penetrate the soil layer because either water movement is largely horizontal, or they have the ability to attenuate diffuse pollutants. Lateral flow from these soils may contribute to groundwater recharge elsewhere in the catchment. They generally have high clay content.	

Each is based on the physical and chemical properties of soil which affect the downward passage of water and contaminants, and include: texture, structure, soil water regime and the presence of distinctive layers such as raw peaty top soil and rock or gravel at shallow depth (EA 2009*c*). The vulnerability of groundwater to pollution is also determined by the type of the aquifer. The EA uses aquifer designations that are consistent with the Water Framework Directive. These designations reflect the importance of aquifers in terms of groundwater as a resource (drinking water supply) and also their role in supporting surface water flows and wetland ecosystems (EA *c*2010). The EA aquifer designations used in the DSS and their descriptions is provided in *Table 6.15*.

Table 6.15 – Aquifer designation and their descriptions (EA 2010c)

1	Principal aquifers (*major aquifers*)	These are layers of rock or drift deposits that have high inter-granular and/or fracture permeability - meaning they usually provide a high level of water storage. They may support water supply and/or river base flow on a strategic scale. In most cases, these are aquifers previously designated as major aquifer.
2	Secondary aquifers	These include a wide range of rock layers or drift deposits with an equally wide range of water permeability and storage. Secondary aquifers are subdivided into two types:
2*a*	Secondary A (*minor aquifers*)	Permeable layers capable of supporting water supplies at a local rather than strategic scale, and in some cases forming an important source of base flow to rivers. These are generally aquifers formerly classified as minor aquifers.
2*b*	Secondary B (*non-aquifers*)	Predominantly lower permeability layers which may store and yield limited amounts of groundwater due to localised features such as fissures, thin permeable horizons and weathering. These are generally the water-bearing parts of the former non-aquifers.
2*c*	Secondary Undifferentiated, U (*variable aquifers*)	In some cases where it is not possible to attribute either category A or B to a rock type. In most cases, this means that the layer in question has previously been designated as both minor and non-aquifer in different locations due to the variable characteristics of the rock type.
3	Unproductive strata (*aquitards*)	These are rock layers or drift deposits with low permeability that has negligible significance for water supply or river base flow.

6.6.3 Generic quantitative risk assessment

The DSS generic quantitative risk assessment involves comparing measured site sample concentrations with GAC values. Site samples are added into the project using the form in *Figure 6.18*. Two buttons are provided at the end of the form, one resetting the form and the other for adding the samples and their information to the project. The results of the chemical analysis of site samples collected are compared with GAC values, which could be EA Soil Guideline Values (SGV) or the Dutch Intervention Values (DIV) (*Fig 6.17*).

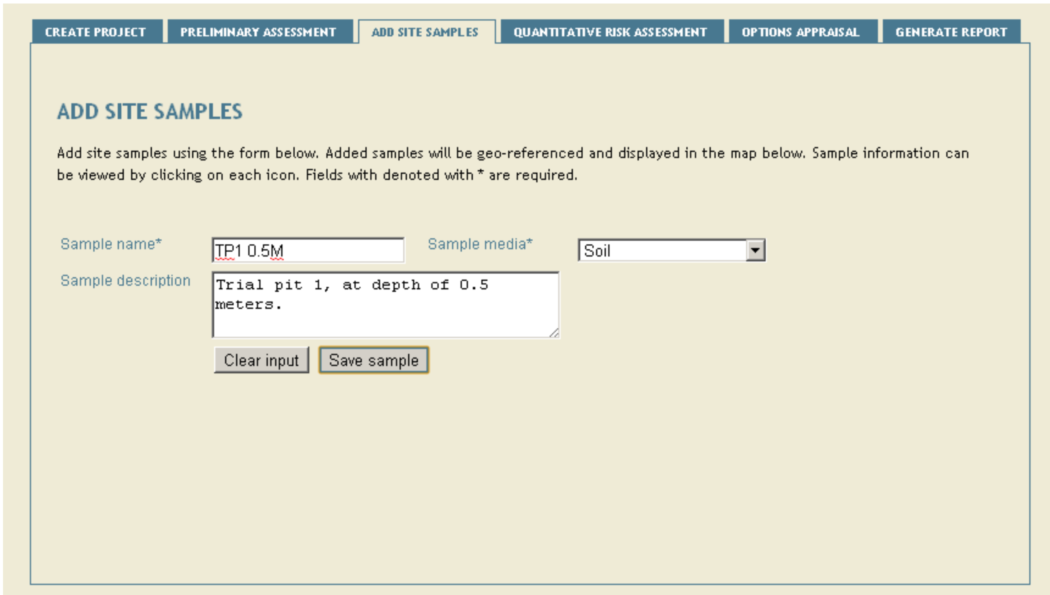

Figure 6.17 – Adding site samples to the decision support system

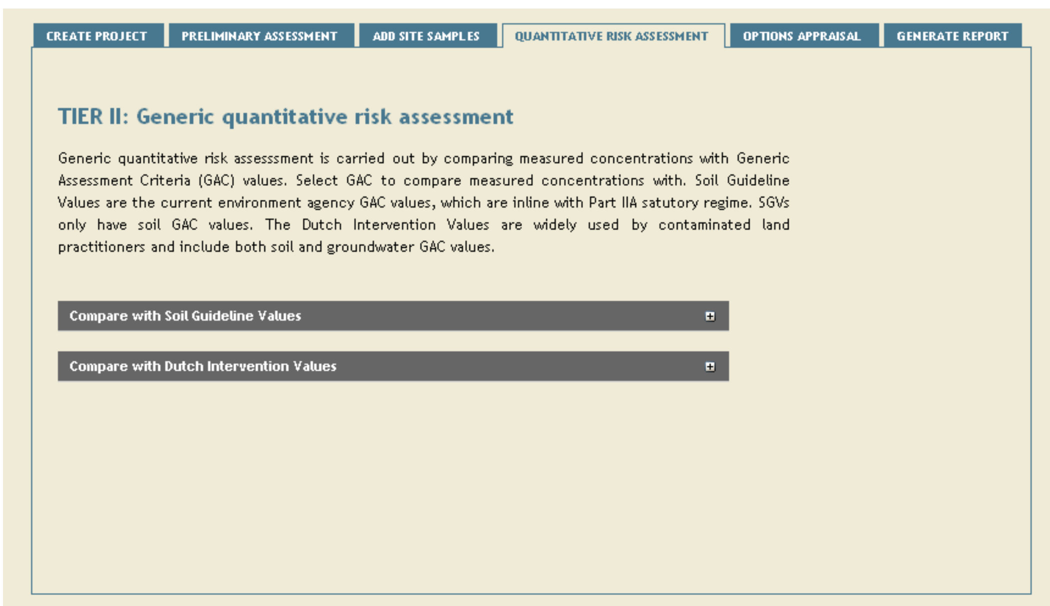

Figure 6.18 – Generic quantitative risk assessment of the decision support system

Two sets of GAC values have been provided because although the current EA GAC values

(SGVs) have been in use since 2002 and old values withdrawn, most contaminated land practitioners still use the withdrawn GAC values, the Inter-Departmental Committee for the Redevelopment of Contaminated Land (ICRCL) GAC values. There is only one ICRCL GAC value for petroleum hydrocarbon contaminants, and practitioners commonly use the DIVs. The derivation of the GAC values is provided in Appendix I. The input forms for comparison with SGVs and DIVs are provided in Figures 6.19, 6.20 and 6.21 respectively.

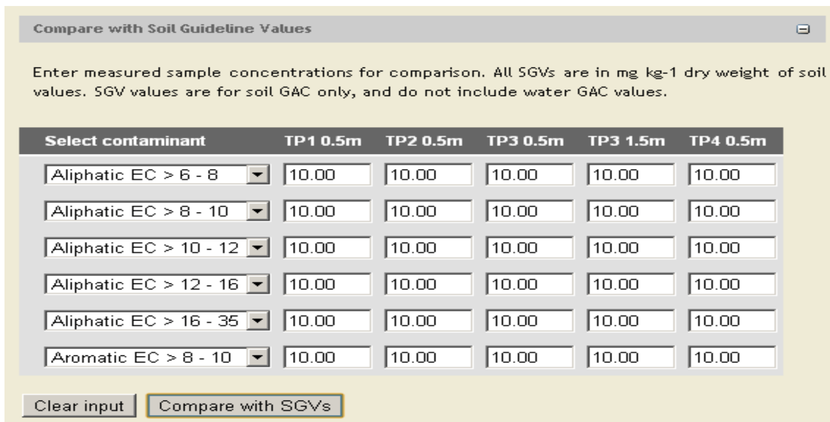

Figure 6.19 – Comparing measured sample concentrations with soil guideline values

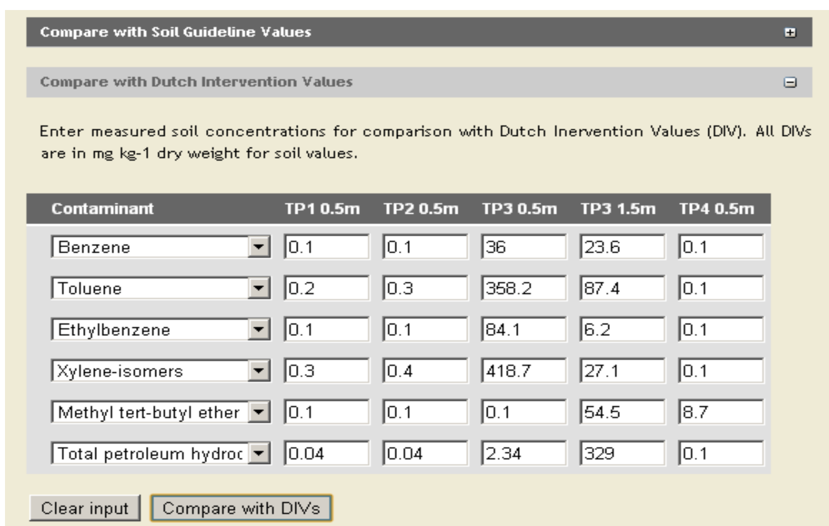

Figure 6.20 – Comparing measured soil sample concentrations with Dutch intervention values

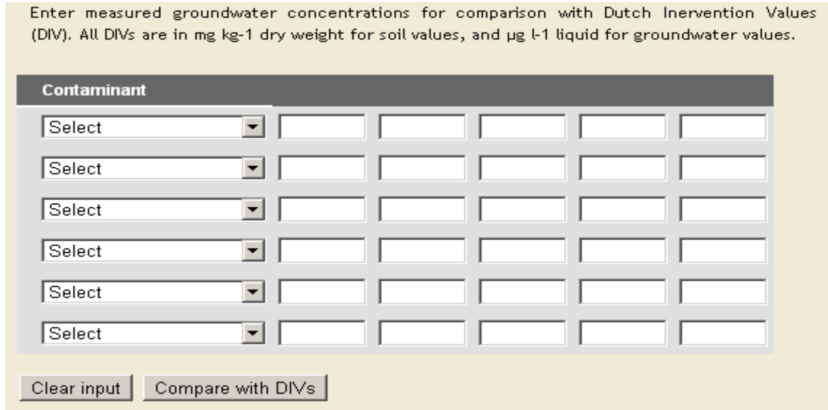

Figure 6.21 – Comparing measured groundwater sample concentrations with Dutch intervention values

6.6.4 Remediation design and options appraisal

The final stage of the decision-making process is the appraisal of remediation technologies. This is provided in page five of the DSS (*Fig 6.22*). The options appraisal process involves remediation design by selecting from different parameters, selection of remediation technologies to be compared, selection of sustainability criteria and sub-criteria for assessing the remediation technologies. Input parameters are pre-selected by the DSS based on input provided by users in previous sections. These can be edited by the decision maker(*s*) based on the management objectives and the decision maker(s) preferences using the menu choices on the left hand side of the page, which include specific soil zone contaminated the relative budget and time, amount of waste by-product and transport. Although the criteria are similar to the sustainability sub-criteria used in the decision model, the decision model only ranks remediation technologies based selected criteria, and does not select remediation technologies. The output of the DSS is a site report generated from the inputs provided for each project.

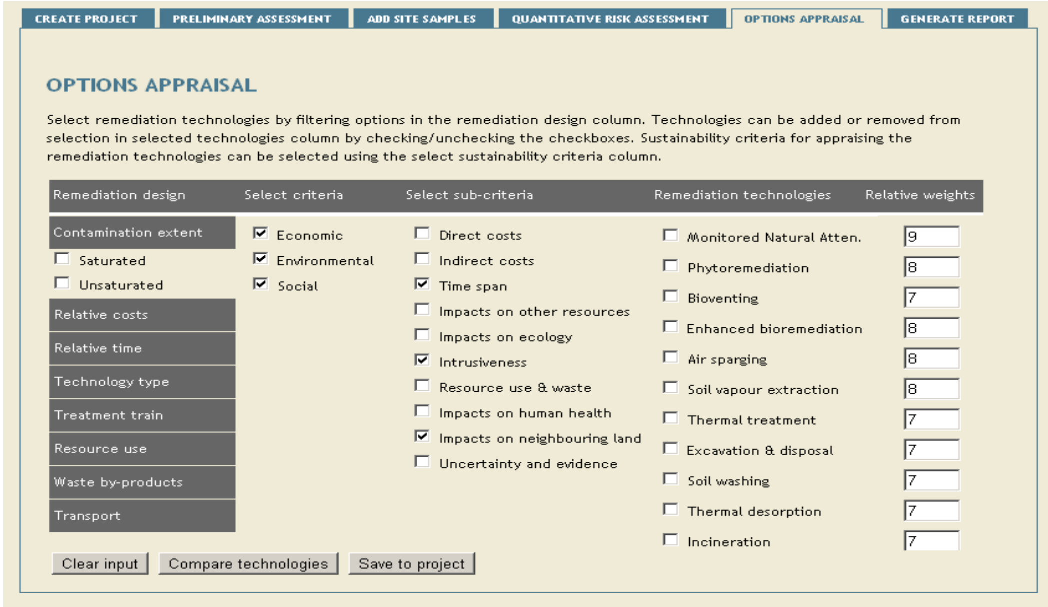

Figure 6.22 – Remediation design and options appraisal of the DSS

Figure 6.23 – Output of the DSS

6.7 CONCLUSION

This chapter presented the development of a Web-based knowledge-based DSS for the sustainable management of contaminated land. The developed DSS is intended to provide an integrated framework for supporting contaminated land management decision-making using the development framework presented in *chapter 5*. A number of DSS for the integrated management of contaminated sites exist, however none of these DSS can be applied within the context of the current UK contaminated land regime however, and none of these explicitly addresses human health issues from land contamination. With respect to remediation design and options appraisal, none of these DSS explicitly addresses the sustainability of the remediation technologies.

The DSS has been developed for the risk assessment of human health and the sustainable management of petroleum hydrocarbon contamination using the current UK contaminated land regime and supporting guidance and technical reports. The DSS consists of three core components: (*i*) a database component containing the DSS database; (*ii*) a model component containing a decision model; *and* (*iii*) a UI component, which provides the architectural framework for integrating the different DSS components. A DSS for petroleum hydrocarbon contamination has been developed for a number of reasons: (*i*) time limitation of the project; (*ii*) extent of petroleum hydrocarbon contamination in the UK; *and* (*iii*) the maturity of the knowledge-base and issues regarding petroleum hydrocarbon contamination. However the DSS has been developed in such a way as to permit easy re-use of any of its components, *i.e*. its database, decision-model and UI components; and/or the integration of other elements to existing components.

The developed DSS consists of three key stages covering the management decision-making process: (*i*) preliminary qualitative risk assessment; (*ii*) generic quantitative risk assessment; *and* (*iii*) options appraisal of remediation technologies. The integration of the decision-making process is intended to provide a framework for the rapid risk assessment and management of contaminated land, which should cut assessment and management costs by focussing attention to areas of concern. The developed DSS should also provide consistency, documentation, rationality and transparency to the decision-making process, thereby increasing confidence in the decision outcome.

7 EVALUATION OF DECISION SUPPORT SYSTEM

7.1 INTRODUCTION

An important part of the development of Decision Support Systems (DSS) is its evaluation in order to establish what the DSS knows, knows correctly, and/or what it does not know (O'Leary 1987). The DSS evaluation process typically involves: (*i*) verification, validation and quality control of the usability of the overall system; *and* (*ii*) investigating the assumptions and limitations of the DSS, its appropriate uses and why it produces the results it does (Borenstein 1998). Verification involves testing and debugging the software code, and is typically carried out throughout the development process, and validation involves testing the appropriateness of the DSS in supporting real world decision problems. Since it is impossible to prove a DSS is a truthful representation of the real world, validation is primarily concerned with demonstrating that the DSS has appropriate underlying relationships to permit an acceptable representation (Finlay 1988).

The developed DSS was evaluating by: (*i*) verification throughout the DSS development process by testing and debugging the source code to ensure there are no errors in the DSS code; (*ii*) independently validating each component with test input data; *and* (*iii*) validating the overall DSS with case studies to see that an acceptable output is achieved for different sets of decision problems. The validation process was carried through bothstructural and functional testing, which involved testing the design and development of the individual components of the DSS and testing DSS inputs and outputs, usually against real life case studies (Sailors et al 1996). This is because functional testing on its own often only evaluate the functionalities needed for solving individual case studies, and may not evaluate all aspects of the DSS or in cases some of the DSS component(s) at all (Finlay 1988, Sailors et al 1996), and structural testing on its own may also be insufficient for testing the interactions between individual elements.

7.2 VERIFICATION OF THE DECISION SUPPORT SYSTEM

The DSS was verified throughout the development process to ensure the source code was standardised and had no 'bugs' and that the syntax of the different languages used in the development of the DSS conform to standards they were written in. MySQL, PHP and CLIPS all have advanced error handling functions for checking syntax as the DSS was being developed, which prevents code from running if it contains any errors. Error handling is used to highlight errors identified in program source codes. The default error handler for PHP, the die ( ) function, was used for detecting all possible error conditions throughout the development of the User Interface (UI) of the DSS. The die ( ) function provided location and information regarding any problems that the code might have.

All the database queries used in the PHP source code were tested in the MySQL command line to ensure they return the desired result(s). Within the PHP code, the mysql_error ( ) function was used for identifying any errors with executing database queries. Syntax validators were used to verify the XHTML and CSS syntax of each page after the DSS development was completed. The validators used were the W3C ® Markup Validation Service for validating the XHTML source code against XHTML 1.0 standards (W3C 2010*a*), the W3C® CSS Validation Service for the style sheets, which checks style sheets against the syntax, properties and values defined in the CSS 2.1 specification (W3C

2010*b*). JSLint, a JavaScript program that looks for problems in JavaScript program, was used for verifying the JavaScript code (JSLint 2010).

The syntax of the knowledge-base code was verified using the inbuilt CLIPS debugging commands. The *watch* command was used to ‘watch’ facts and rules are they were being asserted or retracted, and rules as they were being fired. CLIPS automatically prints a message indicating that an update has been made to the fact list whenever facts are asserted or retracted and when facts are being watched. All fact assertions and retractions are displayed when facts were being watched, and all rule firings are displayed when rules were being watched. The *matches* command and for verifying which patterns in a rule matched facts. The list of rules on the agenda can be displayed using the agenda command.

7.3 VALIDATION OF THE DECISION SUPPORT SYSTEM

Each of the three components of the DSS was independently validated before the integrated DSS was validated. The UI was validated by testing all the input parameters and constraints to ensure defined constraints hold. All the forms in the UI have been developed with inbuilt validation which ensures the DSS accepts only valid input parameters. For example, if an entry is not supplied to a required input field, a form will not be submitted and an error message will be generated to indicate where and what the error is. The database was validated by reverse engineering the database model using a data modelling software, the Toad® Data Modeller (TDM 2010) to generate an Entity Relationship Diagram (ERD) for the database schema. The reverse engineered ERD model was identical to the ERD model developed in the database design stage. All the database dependencies and constraints were also tested. The knowledge-base was validated by calling different rules to check the accuracy and completeness of the knowledge it represents. The independent validation of components is limited at validating different parts of the DSS however, and does not test how the DSS works as an integrated system (O'Leary 1988). In order to adequately test the integrated DSS, real life case studies have been used to validation. Each case study contains information and data relating to preliminary (qualitative) risk assessment; generic quantitative risk assessment and options appraisal of remediation technologies, and the outputs from the DSS were then compared with the expert judgments from the case studies.

Validating input data is an essential requirement for the quality and security of software applications (Brinzarea et al 2006). The form inputs were validated using AJAX validation which takes the advantages of both client- and server-side validation. Client-side validation is processed by the client (web browser) to check that the input values are of the correct data type, length and/or size. Client-side validation is implemented using JavaScript, and is more efficient as it saves time and bandwidth by highlighting errors before the input values are submitted. However most browsers are outdated or have JavaScript disabled and the code can be easily modified or bypassed; therefore client-side validation may not always be sufficient. Server-side validation is implemented using server-side languages like PHP, which checks if input values are correct after the form has been submitted. When a page containing invalid data is sent, an empty form reloads prompting the user to fill the form all over again (Brinzarea et al 2006). AJAX highlights input errors as the form is being filled, and at the same time sends HTTP requests to the server in the background.

7.3.1 Case study 1: Redevelopment of a service station to a domestic dwelling

A fuel filling station has been decommissioned and was being converted to domestic dwelling. During the decommissioning process, detailed risk assessments were carried out by Exeter Environmental Consulting Services (EECS) to assess the potential for soil and groundwater contamination, and to identify possible receptor(s) and their vulnerability, in compliance with requirements of the UK contaminated land regime. The site consisted of a large filling station forecourt surfaced with concrete and asphalt and a general store, and included a former mechanical workshop used for storage. A total of ten fuel tanks were on the sites. Six of the ten fuel tanks were still in use, and the remaining four tanks were redundant, although still in place.

7.3.1.1 Potential pollutant-linkages

The site was in a low density residential area bounded on the north side by a road and on the south side by fields. No other industrial sites were within 500*m* of the site. The soil at the site was graded intermediate with regards to groundwater vulnerability, which implies it could transmit a wide range of pollutants. The site was also overlying an area designated as variably permeable minor aquifer. No abstraction licences are in force within 500*m* of the site. Risk assessment indicated that contamination from the filling station activities was unlikely, given the records on tank testing and inspection. However, if contamination of the soil or near-surface water should occur, geological and groundwater conditions would be likely to encourage migration of any contaminant off site, with two nearby streams and local groundwater potentially being at risk. Leakage from the fuel tanks was the most likely source of contamination by fuel organics into the local environment and the subsequent contamination of sensitive receptors. Surface spillages during tank re-filling and daily business activities could also have lead to the contamination of surface soil layers by washing from the forecourt or draining through surface cracks leading to near-surface contamination. Although the centre section of the forecourt was concrete surfaced, and the areas to each side were tarmac, some gaps between the two surfaces were noted.

7.3.1.2 Sampling and chemical analysis

Further assessment was carried out to ascertain the condition of the soil surrounding the fuel tanks, of any groundwater present and the conditions of the two streams. The tanks were removed prior to sampling, and groundwater was encountered below two tanks. The water below one of the tanks contained droplets of an oily substance, and the water below the other tank showed no visible contamination, however fuel odours were present in this area. The remaining tank pits were dry. The floor of the area previously used as a mechanical workshop appeared sound with little evidence of its previous usage. Soil samples were collected from cavities following tank removal, the workshop, from one trial pit, and just below the base of the tank pits to detect possible contamination in the more porous surface layers. Samples were collected at a depth of between 0.5*m* and 3.5*m*. Samples taken from the deeper levels could be expected to detect any contamination, which may present a risk to groundwater. Surface water samples were collected from the two identified streams and groundwater samples were collected from below the tanks. A total of 12 soil and 4 water samples were collected for the site. The chemical analysis of the samples was conducted by a UKAS and MCERTS[51] accredited laboratory for the relevant

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The *United Kingdom Accreditation Service* (UKAS) is the sole national accreditation body recognised by government to assess, against internationally agreed standards, organisations that provide certification, testing, inspection and calibration services. The *Monitoring Certification Scheme* (MCERTS) is an *Environment Agency*'s (EA) certification framework that covers a range of monitoring, sampling and suites of key contaminants identified (*Tables 7.1* and *7.2*). The measured sample concentrations were compared with Dutch Intervention Values (DIVs) as Generic Assessment Criteria (GAC) values as no UK values were available then (2002).

Table 7.1 – Measured soil sample concentrations compared with the Dutch Intervention Values in mg kg^{-1} dry weight soil

Sample ID	Benzene	Toluene	Ethyl benzene	Xylene (total) isomers	MTBE	TPH
Tanks 1 - 4						
TP1/south/0.6m	<0.001	<0.001	<0.001	<0.001	<0.001	<10
TP1/south/3.5m	<0.001	<0.001	<0.001	<0.001	<0.001	28
TP1/north/0.5m	<0.001	<0.001	<0.001	<0.001	<0.001	87
TP1/north/2.9m	<0.001	<0.001	<0.001	0.007	<0.001	793
Tanks 5 and 6						
TP2/south/0.6m	<0.001	<0.001	<0.001	<0.001	<0.001	137
TP2/south/3.0m	<0.001	<0.001	<0.001	<0.001	<0.001	345
TP2/north/0.6m	<0.001	<0.001	<0.001	<0.001	<0.001	40
TP2/north/3.0m	<0.001	<0.001	<0.001	<0.001	0.035	30
Tanks 7 – 10						
TP3/0.5m	<0.001	<0.001	<0.001	<0.001	<0.001	<10
TP3/2.2m	<0.001	<0.001	<0.001	0.002	0.001	180
TP3/2.6m	0.002	0.011	0.308	1.295	<0.001	1746
Workshop/0.5m	<0.001	<0.001	<0.001	<0.001	<0.001	<10
DIVs (mg kg^{-1})	**1.0**	**130**	**50**	**25**	**~**	**5000**

inspection activities within which environmental measurements can be made in accordance with the current EA quality requirements.

Table 7.2 – Measured groundwater and surface water sample concentrations compared with the Dutch Intervention Values in $\mu g\ L^{-1}$ in solution

Sample ID	Benzene	Toluene	Ethyl benzene	Xylene (total) isomers	MTBE	TPH
North stream	<0.1	0.2	<0.1	0.3	<0.1	<0.04
South stream	<0.1	0.3	<0.1	0.4	<0.1	<0.04
Tank 5 pit	36.0	358.2	84.1	418.7	54.5	2.34
Tank 6 pit	23.6	84.7	6.2	27.1	8.7	329
DIVs ($\mu\ L^{-1}$)	**30**	**1000**	**150**	**70**	**~**	**600**

7.3.1.3 Risk assessment findings

All collected samples were analysed for *Benzene*, *Toluene*, *Ethylbenzene*, *Xylenes* (BTEX); *Methyl Tert-Butyl Ether* (MTBE) and *Total Petroleum Hydrocarbons* (TPH). The measured sample concentrations were compared with DIVs. Measured soil samples indicated fairly low levels of contamination for all contaminants tested. MTBE was the highest level of contaminant detected which was at TP2/north/3.0*m*. The highest levels of soil contamination were noted in the central forecourt area but none exceeded the DIV. High concentrations of fuel products were also found in groundwater to the east of the site in TP1 (*tanks 5 and 6*) with some benzene and xylene exceeding guideline values. No recommendations for remediation have been given at this stage of the management process as more information is required. In order to determine the area of contamination, and in particular to discover if migration off-site has occurred, it is recommended that small observation boreholes are installed around the periphery of the site, from which further soil and water samples can be obtained, and groundwater flow direction can be investigated.

7.3.1.4 The DSS site report

The case study was applied to the results of the DSS were very similar to the findings above, with few expected exceptions. With regards to preliminary risk assessment and site characterisation, the DSS result is generally comparable to the findings of the risk assessment above. The results of the preliminary risk assessment are given below. The DSS only provides judgements on input parameters provided:

- Land end-use is different from current land use. A planning permit will be required, and the planning and/or building regulations may apply. Part IIA EPA (1990) definition of contaminated land applies.
- Some neighboring land end use not the same as land end use. Generic Assessment Criteria values for more sensitive land use than site end-use might be required for quantitative risk assessment.
- High soil leaching potential could be a possible pathway to surface water resources.
- Secondary (U) Undifferentiated aquifers are variable aquifers that may have the characteristics of both secondary A and secondary B aquifers. High 3 soil leaching potential with secondary U aquifer is likely to pose a MODERATE-LOW risk of pollution of stored and/or controlled water resources.

Generic Quantitative Risk Assessment (GQRA) of the DSS, which involves comparing measured site sample concentrations with GAC values (*Figs 7.1 –7.4*), accurately and indicated all measured soil sample concentrations were below the GAC threshold, with only the water sample at trial pit 5 exceeding values for benzene and total xylene isomers. The DSS GQRA result output is: '*at least one sample concentration exceeds GAC value for* *land end-use. Further action might be necessary'.*

1 2 3 next

Contaminant	GAC	TP1/0.6m	TP1/3.5m	TP1/0.5m	TP1/2.9m	TP2/0.6m
Benzene	1.00	0.001	0.001	0.001	0.001	0.001
Toluene	130.00	0.001	0.001	0.001	0.001	0.001
Ethylbenzene	50.00	0.001	0.001	0.001	0.001	0.001
Xylene-isomers	25.00	0.001	0.001	0.001	0.007	0.001
MTBE	100.00	0.001	0.001	0.001	0.001	0.001
TPH	5000.00	10.000	28.000	87.000	793.000	137.000

Figure 7.1 – GQRA result for measured soil concentrations 1

prev 1 2 3 next

Contaminant	GAC	TP2/3.0m	TP2/0.6m	TP2/3.0m	TP3/0.5m	TP3/2.2m
Benzene	1.00	0.001	0.001	0.001	0.001	0.001
Toluene	130.00	0.001	0.001	0.001	0.001	0.000
Ethylbenzene	50.00	0.001	0.001	0.001	0.001	0.000
Xylene-isomers	25.00	0.001	0.001	0.001	0.001	0.000
MTBE	100.00	0.001	0.001	0.035	0.001	0.000
TPH	5000.00	345.000	40.000	30.000	10.000	0.000

Figure 7.2 – GQRA result for measured soil concentrations 2

prev 1 2 3 next

Contaminant	GAC	TP3/2.2m	TP3/2.6m	WS/0.5m		
Benzene	1.00	0.001	0.002	0.001		
Toluene	130.00	0.000	0.000	0.000		
Ethylbenzene	50.00	0.000	0.000	0.000		
Xylene-isomers	25.00	0.000	0.000	0.000		
MTBE	100.00	0.000	0.000	0.000		
TPH	5000.00	0.000	0.000	0.000		

Figure 7.3 – GQRA result for measured soil concentrations 3

Contaminant	GAC	North stream	South stream	Tank 5 pit	Tank 6 pit
Benzene	30.00	0.100	0.100	36.000	23.600
Toluene	1000.00	0.200	0.300	358.200	84.700
Ethylbenzene	150.00	0.100	0.100	84.100	6.200
Xylene-isomers	70.00	0.300	0.400	418.700	27.100
Methyl	9200.00	0.100	0.100	54.500	8.700
Total	600.00	0.040	0.040	2.340	329.000

Figure 7.4 – GQRA result for measured groundwater concentrations

Although the case study involved only risk assessment and has no judgement or recommendation on the possible remedial action, the options appraisal component was used derive the relative ranking the DSS remediation technologies based on different assumptions: (*i*) since the site was being converted into residential dwellings, the remediation time span is a huge factor as there will be indirect costs associated with capital loss the longer the land remained contaminated or within the remediation life-cycle; (*ii*) cost is also a major deciding factor; *and* (*iii*) contaminated land regulations require the remediation process to be protective of human health and controlled waters at minimum. Therefore the direct cost, indirect cost, time span, impacts on other resources and impacts on human health sustainability sub-criteria were selected. The relative ranking of the remediation technologies available based on the selected criteria is:

Table 7.3 – Relative ranking of remediation technologies with respect to intrusiveness

Technology	Certainty value
Monitored natural attenuation	20
Phytoremediation	19
Bioventing	18

Enhanced bioremediation	20
Air sparging	17
Soil vapour extraction	17
Thermal treatment	16
Excavation and disposal	16
Soil washing	18
Thermal desorption	16
Incineration	14

The highest scoring sub-criteria are: (*i*) direct cost; *and* (*ii*) impacts on other resources. These were selected individually to determine their independent rankings respectively:

Table 7.4 – Relative ranking of remediation technologies with respect to direct costs

Technology	Certainty value
Monitored natural attenuation	7
Phytoremediation	7
Bioventing	4
Enhanced bioremediation	4
Air sparging	3
Soil vapour extraction	5
Thermal treatment	2
Excavation and disposal	6
Soil washing	1
Thermal desorption	1
Incineration	1

Table 7.5 – Relative ranking of remediation technologies with respect to impacts on other resources

Technology	Certainty value
Monitored natural attenuation	10
Phytoremediation	9
Bioventing	7
Enhanced bioremediation	7
Air sparging	8
Soil vapour extraction	6
Thermal treatment	4
Excavation and disposal	2
Soil washing	5
Thermal desorption	3
Incineration	1

Adding other sustainability criteria or removing from the selected five above will also have an effect on the ranking of the remediation technologies. The remediation design column can also be used to filter the remediation technologies based on site specific requirements.

7.3.2 Case study 2: Redevelopment of a garage workshop and joinery business for domestic dwelling

The second case study applied to the DSS is of a risk assessment at a redevelopment site in support of a planning application, which was also carried out by EECS. Adjacent to the southern boundary and sharing the current access from east is a site also currently undergoing redevelopment to a domestic dwelling. A commercial garage, now closed, is located at the south-western corner. The site abuts public roads to the north and east. The site had been cleared at the time of the ground survey, however previously existing buildings included a large joinery workshop built of reinforced concrete and timber, a private garage/workshop, and wooden shed. The most recent potentially contaminative use of the site was by the joinery business and private garage/workshop.

7.3.2.1 Potential pollutant-linkages

The site is located on a minor aquifer of variable permeability, however no groundwater was found on site in trial pits to depths of 2.3*m*, at which depth the underlying shale is only slightly weathered. The nearest surface water feature is located approximately 170*m* to the north-west. It is likely that if any contaminants are found on site, the surface water receptor could be most at risk due to topographical flow. No licensed groundwater abstractions exist within 2*km* of the site. The soil is classified as having intermediate leaching potential, indicating that it can readily transmit non-adsorbed pollutants and discharges, but have some ability to attenuate due to their large clay or organic matter contents. No fuel is known to have been stored on site, and a recent storage of fuel nearby is considered to be the primary potential source of sub-surface contamination, due to leakage through tank corrosion, or from the delivery systems. Potential key contaminants have been identified to include aromatic hydrocarbons and Polyaromatic Hydrocarbons (PAH). During development, potential contaminants if present, could be mobilised through disturbance of the soil surface, and thereby pose an inhalation risk to both the workforce and passers-by. Contaminated soils also pose a potential risk to the future occupiers of the dwellings through use of the open spaces, either through inhalation, adsorption or ingestion.

7.3.2.2 Sampling and chemical analysis

A sampling strategy that focussed on areas which are destined to become gardens or accessible open spaces, while remaining representative of the site as a whole was used. Thus the strategy could be expected to detect likely hot-spots and to cover the most sensitive end use areas. Trial pits were excavated to below the anticipated foundation depth (1.4 – 1.6mBGL). Samples were collected at 0.5*m* depth in four trial pits to assess surface soil conditions, and additionally at 1.5*m* depth in the trial pit immediately down-slope of the tanks at one trial pit, in order to intercept possible hydrocarbon migration at depth. As no groundwater was detected on site, no groundwater samples were collected. Analysis was conducted by a UKAS and MCERTS accredited laboratory for the relevant suites of key contaminants defined identified (*Table 7.6*). The measured sample concentrations were compared with EA-based SGVs as GAC values.

Table 7.6 – Measured soil sample concentrations for allotment land use for different land use types based on sandy soils at pH 7.0 and SOM of 5% in mk kg^{-1} dry weight soil

Carbon Range	**SGV**	**TP 10.5*m***	**TP 2 0.5*m***	**TP 3 0.5*m***	**TP 3 1.5*m***	**TP 4 0.5*m***
Organic Matter %		1.9	3.8	0.6	0.6	11.5
Aliphatic Ranges Cn – Cn						
EC > 6 – 8	NA	<10	<10	<10	<10	<10
EC > 8 – 10	22.70	<10	<10	<10	<10	<10
EC > 10 – 12	7.00	<10	<10	<10	<10	<10
EC > 12 - 16	40.10	<10	<10	<10	<10	<10
EC > 16 – 21	163.00	<10	<10	<10	<10	<10
EC > 21 – 36	16300.00	<10	<10	<10	<10	466
Aromatic Ranges Cn – Cn						

EC > 6 – 8	NA	<10	<10	<10	<10	<10
EC > 8 – 10	5.30	<10	<10	<10	<10	<10
EC > 10 – 12	9.44	<10	<10	<10	<10	<10
EC > 12 - 16	10.70	<10	<10	<10	<10	<10
EC > 16 – 21	133.00	<10	<10	<10	<10	<10
EC > 21 – 36	157.00	<10	<10	<10	<10	<10

7.3.2.3 Risk assessment findings

The results of the comparisons of the measured soil concentrations with GAC values indicated that the soil was not significantly contaminated, as no target concentrations of the parameters analysed were detected at any sample point except at one trial pit, where aliphatic compounds in the carbon ranges 21 - 36 were detected, the level of which is well below the GAC values. Trial pit observations show that this is a very localised area and not representative of the remainder of the site. The highest risk to receptors was thought to be to human health during the development of the site if the area around trial pit at which the aliphatic compounds was detected is disturbed otherwise no risk to human health is likely. Controlled water receptors were thought to be at a very low risk due to the small volume of contaminated area, the level of contaminants present, the presence of attenuating soils, as well as dilution and dispersion of contaminants prior to reaching either the minor aquifer directly below the site or the surface water issues nearly two hundred metres down-gradient. The sensitivity of these receptors can be deemed as low as no licensed groundwater abstraction points are located within 2*km* of the site and any surface water abstractions within 2*km* of the site is not taken from the stream. It was therefore recommended that no remediation is required unless the area around the trial pit with the detected aliphatic compounds is disturbed during redevelopment works. The remediation of this area can be successfully carried out by removal of the made ground, and replaced with inert material.

7.3.2.4 The DSS site report

This case study was applied to the DSS and the results of the DSS were very similar to the findings above, with few expected exceptions. With regards to preliminary risk assessment and site characterisation, the DSS result is generally comparable to the findings of the risk assessment above. The results of the preliminary risk assessment are given below. The DSS only provides judgements on input parameters provided:

- Land end-use is different from current land use. A planning permit will be required, and the planning and/or building regulations may apply. Part IIA EPA (1990) definition of contaminated land applies.
- Some neighboring land end use not the same as land end use. Generic Assessment Criteria values for more sensitive land use than site end-use might be required for quantitative risk assessment.
- Intermediate soil leaching potential could be a possible pathway to surface water resources. Intermediate 2 soils can possibly transmit weakly or non-adsorbed pollutants and liquid discharges, but are unlikely to transmit adsorbed pollutants.
- Secondary (U) Undifferentiated aquifers are variable aquifers that may have the characteristics of both secondary A and secondary B aquifers. There is likely a LOW risk of pollution of stored and/or controlled water resources.

The GQRA, which involves comparing measured site sample concentrations with GAC values, was also comparable (*Fig 7.5*). In the GQRA from the case study, the results of the comparisons with GAC values indicated that soils were not significantly contaminated, as no target concentrations of the parameters analysed were exceeded at any sample point except at TP4. Aliphatic compounds in the carbon ranges 21 - 36 were detected at TP4, which is well below the GAC values.

Contaminant	GAC	TP1 0.5m	TP2 0.5m	TP3 0.5m	TP3 1.5m	TP4 0.5m
Aliphatic EC > 6 - 8	22.70	10.000	10.000	10.000	10.000	10.000
Aliphatic EC > 8 - 10	7.00	10.000	10.000	10.000	10.000	10.000
Aliphatic EC > 10 - 12	40.10	10.000	10.000	10.000	10.000	10.000
Aliphatic EC > 12 - 16	163.00	10.000	10.000	10.000	10.000	10.000
Aliphatic EC > 16 - 35	16300.00	10.000	10.000	10.000	10.000	466.000
Aromatic EC > 8 - 10	5.30	10.000	10.000	10.000	10.000	10.000
Aromatic EC > 10 - 12	9.44	10.000	10.000	10.000	10.000	10.000
Aromatic EC > 12 - 16	10.70	10.000	10.000	10.000	10.000	10.000
Aromatic EC > 16 - 21	133.00	10.000	10.000	10.000	10.000	10.000
Aromatic EC > 21 - 35	157.00	10.000	10.000	10.000	10.000	10.000

Figure 7.5 – GQRA results for measured soil concentrations

The DSS GQRA however shows all measured sample concentrations compared with allotment land use GAC are exceeded. This is because the case study risk assessment uses specific Soil Organic Matter (SOM) value, which was different at each sampling point, where as the DSS uses a generic SOM value of 5.00% and a pH of 7.0. The DSS was able to compare with the appropriate GAC values however, and compares the measured concentrations with allotment land use GAC values, as land end-use is residential. Although this could be considered conservative and the GAC could be compared with residential land use values, it is not known whether food will be grown on the site, and no values for residential land use with plant uptake exist. The GQRA results conclude with the DSS inferring since at least one GAC value has been exceeded, further action might be necessary on the site. Although the DSS has not explicitly recommended ranking of the technologies, options appraisal can still be carried out for the site using the DSS.

7.3.3 Case study 3: Removal of contaminated land from residential gardens

The main objective of the management of this hydrocarbon contaminated site was to remove the contaminated material from the garden areas of houses built upon a former town gas site to a risk assessed level and reinstate existing landscaping by a geo-environmental company, Soilutions. The specific requirements of the project were: (*i*) minimal disruption to occupants of neighbouring houses; (*ii*) protection of the driveways and surrounding hard landscaping; *and* (*iii*) the completion of the works within a two week period to meet with financial year end date. The remediation technology selected by the practitioners was excavation and disposal, because it best fit the project requirements. The same set of criteria was applied to the DSS to validate the knowledge-base.

The criteria selected in the DSS options appraisal were: (*i*) time-span; (*ii*) impacts on neighbouring land use; *and* (*iii*) intrusiveness. The relative ranking of the remediation technologies based on these criteria is presented in *Tables 7.7 – 7.9*. The highest ranking remediation technology with respect to 'time-span' criteria is excavation and disposal (*Table 7.7*), and with respect to 'intrusiveness' criteria is monitored natural attenuation (*Table 7.5*). The relative ranking with of both technologies with respect to impacts on neighbouring land is equal. The relative ranking of the technologies with respect to all three criteria is provided in *Table 7.9*. The DSS remediation design tool was used to filter the remediation technologies. The contamination zone (unsaturated) and relative time (less than 0.5 years) options were selected, which filtered the remediation technologies to only relevant ones (*Fig 7.6*). Based on the ranking of the remediation technologies in *Table 7.9* any of the four remaining remediation technologies could have been appropriate (*Fig 7.6*). This could be further reduced by selecting other remediation design options from the left hand side menu to select different remediation technologies.

Table 7.7 – Relative ranking of remediation technologies with respect to time-span

Technology	Certainty value
Monitored natural attenuation	1
Phytoremediation	1
Bioventing	2
Enhanced bioremediation	4
Air sparging	2
Soil vapour extraction	2
Thermal treatment	3
Excavation and disposal	5
Soil washing	4
Thermal desorption	4
Incineration	4

Table 7.8 – Relative ranking of remediation technologies with respect to intrusiveness

Technology	Certainty value
Monitored natural attenuation	7
Phytoremediation	6

Bioventing	4
Enhanced bioremediation	3
Air sparging	5
Soil vapour extraction	5
Thermal treatment	3
Excavation and disposal	1
Soil washing	2
Thermal desorption	2
Incineration	2

Table 7.9 – Relative ranking of remediation technologies with respect to time-span, intrusiveness and impacts on neighbouring land

Technology	Certainty value
Monitored natural attenuation	9
Phytoremediation	8
Bioventing	7
Enhanced bioremediation	8
Air sparging	8
Soil vapour extraction	8
Thermal treatment	7
Excavation and disposal	7
Soil washing	7
Thermal desorption	7
Incineration	7

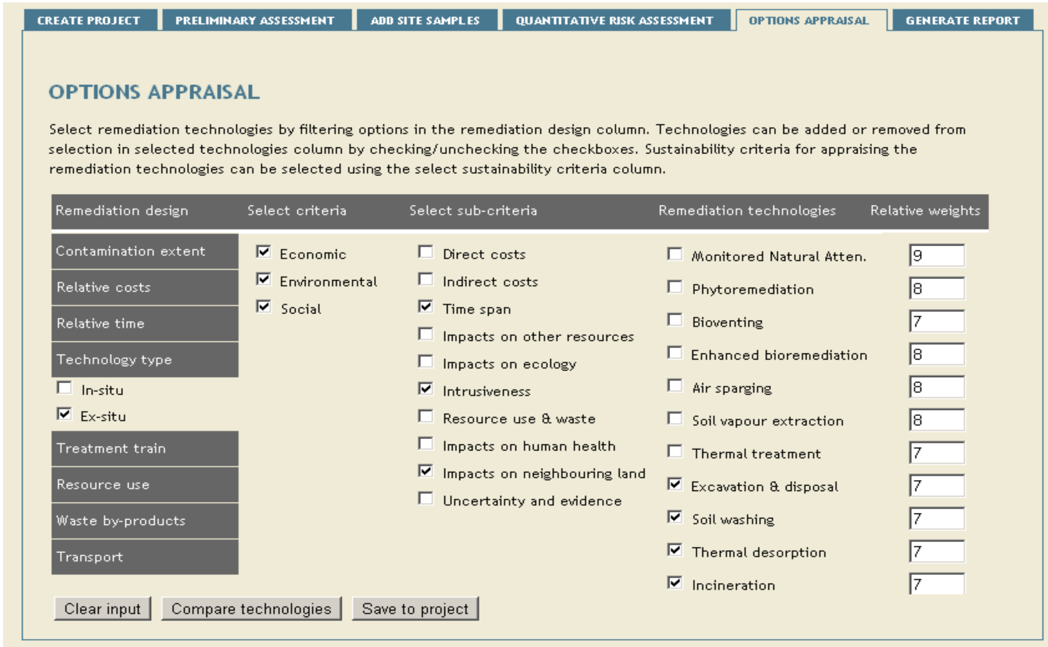

Figure 7.6 – The decision support system selected remediation technologies

7.4 CONCLUSION

The validation of DSS is as critical as its development to ensure adequate performance in real world applications. Yet few works are devoted to this aspect of DSS development to ensure adequate performance in real applications (Sànchez-Marrè et al 2008). The developed DSS has been evaluated by verifying all the source code has been written according to standard and that they contained no bugs. This was carried out throughout the development process. Each of the components of the DSS has been rigorously validated independently. The overall DSS has also been validated using real life case studies in order to establish what the DSS knows, knows correctly, and/or what it does not know (O'Leary 1987). The results of the case studies strongly match the results the DSS produces for all the decision-making tasks, with very few anomalies. The anomalies were expected because of differences in formats of the case studies, and the level of detail of the case studies.

8 CONCLUSION

8.1 SUMMARY AND CONCLUSION

Land contamination is a major environmental and infrastructural problem in industrial countries, with potential detrimental effects on human health, valuable water resources, sensitive ecological systems, property and infrastructure. The effective management of contaminated land typically requires multi-agency regulation and multidisciplinary expertise, involving the integration of vast multidisciplinary knowledge-bases into a coherent decision-making framework within a relevant regulatory context. The management process is complex and is typically undertaken using a phased approach with explicit considerations of risk at each phase of the decision-making process (Hester and Harrison 2001). In the UK the management of contaminated land is undertaken using a tiered risk-based approach (EA 2004*b*), with each incremental tier involving increasing detail and complexity, involving: (*i*) preliminary risk assessment which desk study and site investigation; (*ii*) generic quantitative risk assessment which involve the chemical analysis of site samples and comparing the measured sample concentrations with Generic Assessment Criteria (GAC); *and* (*iii*) detailed quantitative risk assessment which uses site specific assessment criteria (DEFRA 2008). If the outcome of risk assessment requires further action, a risk management strategy is developed and implemented.

Increasingly the goal of remediation is the sustainable management of the contamination, involving: (*i*) either full or partial treatment; (*ii*) isolation; *or* (*iii*) removal of contaminants. Sustainable management involves balancing inevitable trade-offs between competing economic, environmental and social criteria, with ideal (*i.e*. sustainable) solutions aiming to minimise total operational costs, minimise environmental impacts and maximise social benefits. This ideal is rarely achieved on the basis on scientific evidence alone, and increasingly decision-making techniques and Decision Analysis (DA) methods are used to support with balancing trade-offs between decision criteria. It is also possible that a solution that appears suitable and is sufficient and proportional to land end-use may not be feasible technically or economically. Formalised and structured methods like Multi Criteria Decision Analysis (MCDA) provide a means of evaluating these multiple criteria. MCDA decision models have been widely used to support contaminated land decision-making, and have been shown to offer significant improvements in the decision-making process (Bridges et al 2006, Linkov et al 2006b). These MCDA decision models are increasingly being encapsulated into Decision Support Systems (DSS) to automate the solution of the same type of decision problems.

Although numerical and statistical simulation models have long been used to garner insights into contaminated land problems, the complexity of contaminated land management decision problems require the application of new methods. DSS are amongst the most promising solutions because of their ability to integrate different frameworks, architectures, tools and methods for solving high level complexity (Poch et al 2003). DSS allows the relatively easy integration of disciplines from classical fields of all kinds of optimisation, to stochastics, decision theory, decision-making, decision support and so forth (Radermacher 1994). Many DSS have been developed for contaminated land decision support with varying degrees of success in practical applications (CLARINET 2002). However a lot of the DSS are different models integrated to better visualise data or describe systems, and do not specifically address decision problems or help decision makers in making inevitable tradeoffs (Giove et al 2008). The majority of these DSS focus on risk assessment, technology selection and stakeholder involvement (Agostini and Vaga 2008), and rarely look at the overall contaminated land management process holistically. As all aspects of the management process are related and have a bearing on the final decision outcome, there is a need to integrate the different models, software and tools into single portal for effective management.

Many frameworks for developing both generic and discipline specific DSS have been proposed. However no single framework dominates and the development of DSS is still an *ad hoc* process. This is mainly due to the fact that the development of the DSS is a multidisciplinary process involving knowledge of the DSS application area and techniques and tools from a wide range of disciplines. This thesis presented a framework for the development of contaminated land management DSS, taking into account: (*i*) contaminated land management decision-making process and its constraints: (*ii*) the underlying multidisciplinary information required for contaminated land management decision-making; (*iii*) the range of management decisions that can be made; (*iv*) the different policy contexts; *and* (*v*) evolution of the DSS resulting from the changes in the underlying scientific and technical understanding of land contamination. The framework is based on the component-based approach to software development which explicitly addresses issues of re-usability by developing different parts of the DSS as independent components.

The framework was used to develop a problem-specific DSS for the sustainable management of petroleum hydrocarbon contamination, using the current UK contaminated land regime and supporting guidance, regulations and technical reports from other UK Government agencies and organisations. A DSS has been developed as an integrated system for the whole management life cycle, involving: (*i*) preliminary qualitative risk assessment; (*ii*) generic quantitative risk assessment; *and* (*iii*) options appraisal of remediation technologies and remediation design. The input of the system includes information from desk study, site investigation and the results of the chemical analysis of site samples. The DSS provides a site report at the end of the management decision-making process as output, which can be used as information for decision support. The DSS was developed as a Web-based application, on an *n*-tier client-server architecture with the first tier as the presentation layer (the User Interface (UI)), the second tier the application layer (model component), and the third tier the storage layer (database component).

The DSS was developed on an open source LAMP server. The developed DSS consists of three components: (*i*) a database component; (*ii*) a model component; *and* (*iii*) a User Interface (UI) component. The database component was developed as a Relational Database (RDB), using the international standard database language SQL embedded in MySQL database server. The knowledge-base encapsulates the DSS decision model. An MCDA was developed using the Analytical Hierarchy Process (AHP). The result of the model was encapsulated in the knowledge-base using the CLIPS expert system shell. The UI was developed as a Graphical User Interface (GUI) using mixed language programming with a combination of markup, styling and both client- and server-side languages.

An important part of the development of DSS is its evaluation in order to establish what the DSS knows, knows correctly, and/or what it does not know (O'Leary 1987). The developed

DSS has been evaluated using a number of real life case studies. The evaluation process included both verification and validation of the DSS. Verification of the DSS involved testing and debugging the software code, and was carried out throughout the development process. Validation involved testing the appropriateness of the DSS in supporting real world decision problems. However since it is impossible to prove a DSS is a truthful representation of the real world; validation was primarily concerned with demonstrating that the DSS had appropriate underlying relationships to permit an acceptable representation (Finlay 1988).

The outputs of the DSS for all the case studies were similar to the findings of the case studies, with minor expected exceptions. The exceptions were mainly with comparing measured sample concentrations with GAC values in generic quantitative risk assessment. This was due to the generic nature of the DSS GAC values, which were based on 6.00% Soil Organic Matter (SOM) and pH 7.00 for sandy clay soils for all land use types for the EA-based values, and the 5.00% SOM at pH 7.00 for sandy soils for the Dutch-based GAC values. The GAC values for the case studies were more site specific, and were based on SOM and pH values for each collected sample and the site soil type. The DSS GQRA is able to consistently and accurately compare measured sample concentrations with GAC values based on land end-use or in the case of Dutch values, contaminant media and highlight whether further action might be necessary if they are exceeded.

Many expert and knowledge-based systems have been successfully used to deal with real world problems that conventional programming have been unable to solve, especially when dealing with uncertain and incomplete information. In hindsight, using the CLIPS paradigm wasn't the most efficient programming paradigm for integrating the encapsulating the result of the decision-model, as the CLIPS implementation is best suited to problems with no algorithmic solution. A procedural programming language like PHP or JavaScript would therefore have been more suitable because the result of the decision model is fully known. Moreover, it would have taken a lot less time and effort which would have allowed for the addition of other forms of contamination to the DSS.

8.2 RECOMMENDATIONS FOR FUTURE WORK

To the author's knowledge, the developed DSS presented in this thesis represents a first attempt at an integrated DSS for sustainable management of contaminated land under the current UK contaminated land regime of Part IIA EPA 1990 and supporting guidance, regulations, and technical reports. The integration of the decision-making process is intended to provide a framework for the rapid risk assessment and management of contaminated land, which should cut assessment and management costs by focussing attention on areas of concern. The developed DSS should also provide consistency, documentation, rationality and transparency to the decision-making process, thereby increasing confidence in the decision outcome.

From the literature reviewed in the thesis, it can be concluded that future trends in contaminated land management decision-making and decision support will continue to be a growing area of research. Taking into account the cost, time and other resources required for the development of DSS, the DSS presented in this thesis has been developed in such a way as to permit easy re-use of any of its components, *i.e*. its database, decision-model and UI components; and/or the integration of other elements to existing components. The DSS provides a framework for effective decision-making, on which future amendments could be made. Future work on the DSS can focus on:

- Extending the DSS to include other forms of contamination by developing problem specific databases and integrating them into the existing components.
- The DSS could also be extended to include a login feature. At present, the DSS uses cookies and sessions to ensure the same site details are used in each project. This could present a potential disadvantage in cases where older browsers are used, or where cookies have been disabled, as all site details will be lost once the browser is closed. The login feature added to the DSS will ensure users can always retrieve project details by logging in. This could also be advantageous in cases where a site is being monitored, especially if the project involves the comparison of different sets of measured site concentrations over times.
- Extending the DSS to include site specific quantitative risk assessment. The DSS currently only is able to support with generic assessment, which will not be adequate in cases where specific SOM content, pH value and/or soil type is needed.
- Integrating a model for estimating likely site specific sampling strategies could also be developed and integrated with the DSS. The UK EA has published guidance and procedures for statistical analysis of contaminated soil, within the context of Part IIA EPA 1990 and other land use regulations. Several web-based open source map Application Programming Interfaces (API) exist that could be used with statistical techniques to implement robust sampling strategies based on the UK framework. An attempt was made to include this in the developed DSS using Google Maps API, which could not be completed in time.

- Extending the decision model to consider the effects of climate change on remediation technologies. Both the observed and projected changes of the climate system pose unprecedented challenges on contaminated land and its management practices, especially with increased uncertainty and risks to receptors, contaminant fate and transport and the technological efficacy of some remedial technologies. These changes increase the uncertainty in contaminated land management, and contaminated land remediation and management strategies have to be dealt with under new assumptions for effective and sustainable management.

APPENDICES

APPENDIX I: GENERIC ASSESSMENT CRITERIA

Generic Assessment Criteria (GAC) are a set of scientific based values based on generic assumptions of human behaviour and characteristics of contaminant exposure pathways for a range of different scenarios. GAC are derived and published by authoritative bodies, which in England and Wales is the Environment Agency (EA) as Soil Guideline Values (SGVs). SGVs are GAC values that are used for assessing human health risks arising from long-term and on-site exposure to chemical contamination in soil. The SGVs are based on reasonable generic exposure scenarios for long-term aggregated exposure that are health protective for the vast majority of the UK population (EA 2009). SGVs do not represent the ‘trigger’ for an unacceptable intake and therefore do not explicitly define remediation standards, but can be used as an indication of chemical contamination in soil below which the long-term human health risks are considered to be tolerable or minimal (EA 2009). SGVs are used for Generic Quantitative Risk Assessment (GQRA) in the Decision Support System (DSS) as a means of identifying an area of land, and/or specific contaminants that do not warrant further evaluation.

The GAC values that are used for the GQRA in the DSS are based on published values from different sources: (*i*) the EA SGVs; (*ii*) the Land Quality Management/Chartered Institute of Environment Healt*h* (LQM/CIEH) GAC values; *and* the (*iii*) Dutch Intervention Values for the remediation of soil/sediment and groundwater (DIV 2000). The EA SGVs are published GAC values based on the current UK framework for evaluating human health risks from petroleum hydrocarbons in soils (EA 2005). The petroleum hydrocarbon SGVs are based on the Total Petroleum Hydrocarbon Working Group (TPHCWG) toxicity values, and for heavier hydrocarbon fractions, the American Petroleum Institute (API) values which have been modified to fit with the current UK contaminated land regime. The EA SGVs have been derived by the Contaminated Land Exposure Assessment (CLEA) model (CLEA 2005). The CLEA model estimates exposure to chemicals from soil sources by adults and children living or working on land affected by contamination over long periods of time, and compares this estimate to published Health Criteria Values (HCVs). HCVs are benchmark levels of exposure to chemicals at which level long-term human exposure is tolerable or poses a minimal risk. HCVs differ according to whether they relate to adverse effects that are expected to demonstrate threshold effects (Tolerable Daily Intake, TDI) or effects for which no threshold is assumed (Index Dose, ID) (EA 2009).

TDI is the estimated amount of soil (expressed in *body weight*, bw) that can be ingested daily over a life time without appreciable health risk. IDs represent a dose that poses a minimal risk level from possible exposure (EA 2005), and are derived for contaminants for which a threshold for adverse effects cannot be presumed. Exposure at ID is therefore considered to carry some, albeit minimal and often unquantifiable, level of risk (CLR 9). Indicator compounds are the most frequently occurring at petroleum hydrocarbon contaminated sites, and are often the key risk drivers for remediation. Indicator compounds consist of toxic contaminants exhibiting both non-threshold effects threshold effects (*Table 1*). Non-threshold effects represent contaminant toxicity for which there is no threshold level because any exposure, no matter how small will carry some level of risk. Threshold effects represent the level that needs to be exceeded to produce an adverse effect.

Table 1 – Indicator compounds in the UK approach (EA 2005, EA 2009)

NON-THRESHOLD INDICATOR COMPOUNDS	THRESHOLD INDICATOR COMPOUNDS
benzene [a, b]	toluene [a, b]
benzo[*a*]pyrene [a]	ethylbenzene [a, b]
benz[*a*]anthracene	xylene [a, b]
benzo[*b*]fluoranthene	naphthalene [a]
benzo[*k*]fluoranthene	fluoranthene
chrysene	phenanthrene
dibenz[*a*,*h*]anthracene	pyrene
indeno[1,2,3-*c*,*d*]pyrene	

a = published under the old CLEA model

b = published under the new CLEA model

Hydrocarbon fractions represent mixtures and are used for assessing only threshold effects from petroleum hydrocarbon compounds. Any individual mixture may contain thousands or tens of thousands of different individual compounds, each of which may exhibit different toxicity (EA 2005). The environmental behaviour of each hydrocarbon fraction is therefore complex and affected by a variety of factors including the type of crude oil, its solubility, volatility, temperature, soil type, geological setting *etc* (LQM/CIEH 2009). In the UK approach, hydrocarbon fractions are grouped according to Equivalent Carbon (EC) numbers[52], with each fraction containing compounds with similar environmental properties and therefore having similar fate and transport in the environment. The UK approach

[52] The EC number of a hydrocarbon is related to its normalised boiling point (*b.p.*) normalised to the boiling point on an *n*-alkane series, or its retention time on a non-polar *b.p.* gas chromatographic column (EA 2005).

considers aliphatic and aromatic hydrocarbon fractions separately owing to differences in their toxicity and fate and transport characteristics. Aromatic compounds tend to be more soluble in water and slightly less volatile than aliphatic compounds with similar EC numbers (LQM/CIEH 2009).

Table 2 – Hydrocarbon fractions in the UK approach based on EC number (EA 2005)

ALIPHATIC FRACTIONS	AROMATIC FRACTIONS
> 5 – 6	> 5 – 7
> 6 – 8	> 7 – 8
> 8 – 10	> 8 – 10
> 10 – 12	> 10 – 12
> 12 – 16	> 12 – 16
> 16 – 35	> 21 – 35
> 35 – 44	> 35 – 44
> 44-70	

The EA has only published SGVs for some indicator compounds, but none for hydrocarbon fractions. And although a number of other toxicity values exist, *e.g.* TPHCWG (1986), API (1996), MADEP (2000), WHO (2000), DIV (2000) *etc*, the EA has not formally recommended any toxicological values to be used within the UK approach (LQM/CIEH 2009). For the indicator compounds where no SGVs are available and all hydrocarbon fractions, the LQM/CIEH GAC values are used. The LQM/CIEH GAC values are authoritative GAC values that are widely used by contaminated land practitioners in the UK. These values have been derived from a number of sources, of which priority was given to the EA HCVs. Where no HCV values exist, the TPHCWG toxicity values have been used and for the additional fractions that are not covered in TPHCWG, the API toxicity values have been used (LQM/CIEH 2009). The GAC values for the indicator compounds and their sources used in the DSS are presented in *Table 3*. These are from both the EA SGVs and the LQM/CIEH GAC values. All the GAC used for the hydrocarbon fractions in the DSS are LQM/CIEH GAC values, which were derived using the EA CLEA model.

Table 3 – The GAC for indicator compounds used in the DSS

CONTAMINANT	TOXICITY	SOURCE
benzene	Non-threshold	EA (2009*a*)
benzo[*a*]pyrene	Non-threshold	LQM/CIEH (2009)
dibenzo[*a*,*h*]anthracene	Non-threshold	LQM/CIEH (2009)
toluene	Threshold	EA (2009*b*)
ethylbenzene	Threshold	EA (2009*c*)
xylene, *o*-	Threshold	EA (2009*d*)
xylene, *m*-	Threshold	EA (2009*d*)
xylene, *p*-	Threshold	EA (2009*d*)
fluorine	Threshold	LQM/CIEH (2009)
napthanlene	Threshold	LQM/CIEH (2009)

The EA SGVs are derived using the new approach and are all based on sandy loam soil at pH 7.0 for residential, allotment, commercial /industrial land uses. The new approach uses a SOM of 6% for sandy loam soils because at lower SOM, SGVs may not be sufficiently protective. The SGVs for residential and allotment land uses are based on estimates for young children because they are generally more likely to have higher exposures to soil contaminants. The SGVs are based on consideration of oral, dermal and inhalation routes of exposure. The toxicological effects are considered to be systematic and the combined assessment criteria for oral and dermal pathways are used.

Table 4 – The EA SGVs (mg kg^{-1} DW) for indicator compounds for different land uses as used in the DSS which are based on sandy loam soil at pH 7.0 and SOM 6%

CONTAMINANT	RESIDENTIAL	ALLOTMENT	COMMERCIAL
benzene	0.33	0.07	95
toluene	610	120	4.4×10^3
ethylbenzene	350	90	2.8×10^3
xylene, *o*-	250	160	2.6×10^3
xylene, *m*-	240	180	3.5×10^3
xylene, *p*-	230	160	3.2×10^3

Table 5 – The LQM/CIEH GAC (mg kg^{-1} DW) for indicator compounds for different land uses as used in the DSS which are based on sandy soil at pH 7.0 and SOM of 1, 2, and 5%

CONTAMINANT	RESIDENTIAL *with plant uptake*	RESIDENTIAL *without plant uptake*	COMMERCIAL
benzo[*a*]pyrene	1.09	1.32	29.9
dibenzo[*a*,*h*]anthracene	1.10	1.32	29.9
Fluorine	1.84 E+02	2.70 E+03	5.95 E+04
napthanlene	17.0	33.7	1440

The LQM/CIEH GAC values are derived using the old CLEA model approach and are based on sandy soil at pH 7.0 and SOM of 5% for three land use types. The land uses are the standard land uses in CLEA model. The LQM/CIEH has not published any GAC values for allotment land use, but has published values for the residential land use with plant uptake. These values have been used in the DSS database for allotment land use also, as the GAC values are sufficiently protective of allotment land use. There is very little data available for the toxicity of individual hydrocarbon fractions, as most studies have either investigated the effects of whole products or individual compounds. HCV for threshold behaviour is calculated as a *Tolerable Daily Soil Intake* (TDSI), which is defined as the difference between TDI and background *Mean Daily Intake* (MDI), *i.e*. TDSI = TDI - MDI. The MDI is the average "background intake" to which a population may be exposed, which is expressed in terms of mass of substance per day, *e.g*. mg d^{-1} (EA 2005). Only TDI values for EC > 5 – 6 and EC > 6-8 have been published by the EA. For all the other ECs, the LQM/CIEH TDIs have been derived from TPHCWG values and the API values for heavier fractions. Due to limited information on background exposure through food, drinking water and/or air, the EA takes a precautionary approach for all fractions, and assumes MDI is high in comparison to TDI, and requires that the maximum background exposure possible is 80% of TDI, *i.e*. (MDI = 0.8 x TDI x bw). For each fraction therefore, it has been assumed that TDSI = 0.2 X TDI (LQM/CIEH 2009).

Table 6 – The LQM/CIEH GAC values (mk kg^{-1} DW) for different land use types based on sandy soils at pH 7.0 and SOM of 5% (LQM/CIEH 2009)

Contaminant	Generic Assessment Criteria (mg kg^{-1} dry weight soil)		
	ALLOTMENT	COMMERCIAL	RESIDENTIAL
Aliphatic hydrocarbon fractions			
EC > 5 – 6	6.39 E+00	2.88 E+02	6.38 E+00
EC > 6 – 8	2.27 E+01	1.02 E+03	2.27 E+01
EC > 8 – 10	7.05 E+00	3.17 E+02	7.00 E+00
EC > 10 – 12	4.17 E+01	3.04 E+04	4.01 E+01
EC > 12 – 16	1.87 E+02	3.04 E+04	1.63 E+02
EC > 16 – 35	2.68 E+04	6.27 E+05	1.63 E+04
EC >35 – 44	2.68 E+04	6.27 E+05	1.63 E+04
Aromatic hydrocarbon fractions			
EC > 5 – 6	2.75 E+00	1.21 E+02	2.57 E+00
EC > 6 – 8	3.18 E+00	1.39 E+02	2.85 E+00
EC > 8 – 10	1.16 E+01	5.13 E+02	5.30 E+00
EC > 10 – 12	6.39 E+01	2.60 E+03	9.44 E+00
EC > 12 – 16	2.35 E+02	1.24 E+03	1.07 E+01
EC > 16 – 21	3.62 E+02	9.35 E+04	1.33 E+02
EC > 21 -35	4.04 E+02	9.41 E+03	1.57 E+02
EC >35 – 44	4.04 E+02	9.41 E+03	1.57 E+02
Aromatic and aliphatic hydrocarbons			
EC > 44 – 80	4.04 E+02	9.41 E+03	1.74 E+02

Other toxicity values have also been included almost all contaminated land practitioners still exclusively use them. These include the former UK *Inter-Departmental Committee on the Redevelopment of Contaminated Land* (ICRCL) 59/83 Trigger Concentrations for a

series of substances commonly found in contaminated land (ICRCL 1987) and the Dutch values (2000). Although widely used since 1987, the ICRLC have been withdrawn in 2002, because they are out of date technically and their approach is not in line with the current statutory regime (Part IIA EPA) and associated policy, particularly with assessing of *Significant Possibility of Significant Harm* (SPOSH) to human health, which the regime calls for (DEFRA 2002). The ICRCL trigger concentrations have been replaced by the CLEA package since 2002 (CLEA 2002), which represents the key instruments for generic assessment of the human health risks from land contamination. The CLEA package represent cross-Government consensus on the technical approach to undertaking such assessments and are based on the latest scientific knowledge and thinking (DEFRA 2002).
The ICRCL trigger values consist of *threshold* and *action* trigger concentrations (mg kg^{-1} air dried soil). Concentrations below the *threshold* value are considered to be tolerable. The threshold value is therefore similar in interpretation to the current SGV approach. Above the *action* value, the presence of the contaminant can be regarded as undesirable or even unacceptable, so remedial action is unavoidable. Action may need to be taken with concentrations between the threshold and action values.

The Dutch values are used for assessing soil and groundwater where no UK values are available. The ICRCL has only published trigger concentrations for one petroleum hydrocarbon contaminant, the poly-aromatic hydrocarbon. The set of GAC values provided in the DSS is therefore from the Dutch values. Although similar in interpretation to ICRCL values, the Dutch values differ from the UK values in that the Dutch approach uses the *principle of multi-functionality, i.e*. cleaning the land for all land uses. The GAC values used in the DSS from Dutch values are the intervention values because they are closer to interpretation to the current EA SGVs. The intervention values represent the maximum tolerable concentration above which remediation is required.

Table 7 – The Dutch Intervention Values for remediation of petroleum hydrocarbons in soils (mg kg-1 dry weight) and groundwater (µg/l in solution)

Contaminant	Soil/sediment		Groundwater	
	Target value	Intervention value	Target value	Intervention value
benzene	0.01	1	0.2	30
toluene	0.01	130	7.0	1, 000
ethylbenzene	0.03	50	4	150
total xylene isomers	0.1	25	0.2	70
benzo(*a*)anthracene	-	-	0.0001	0.5
benzo(*a*)pyrene	-	-	0.0005	0.05
chrysene	-	-	0.003	0.2
fluoranthene	-	-	0.003	1
indeno(*1,2,3 –c,d*)pyrene	-	-	0.0004	0.05
naphthalene	-	-	0.01	70
phenanthrene	-	-	0.003	5
PAH (sum of 10) [53]	1	40	-	-
methyl tert-butyl ether (MTBE)	-	100	-	9, 200
total petroleum hydrocarbons (TPH)	-	5, 000	-	600

[53] PAH (sum of 10) is the total of anthracene, benzo(a)anthracene, benzo(k)fluoroanthene, benzo(a)pyrene, chrysene, phenantrene, fluoroanthene, indeno(1,2,3-cd)pyrene, naphthalene and benzo(ghi)perylene.

APPENDIX II: DESCRIPTION OF THE REMEDIATION TECHNOLOGIES

The remediation technologies used in the DSS are described below in *Table 1* below. The technologies include *in-situ* and *ex-situ* technologies. In-situ technologies are used for cleaning up the contamination ‘in place’, without removing the contaminated media. Ex-situ technologies involve the excavation of the contaminated media, either for off-site disposal or for on-site treatment, which is then returned into the environment. Both in-situ and ex-situ technologies are grouped according to their treatment mechanisms into: (*i*) biological treatments which use microbes to degrade or transform contaminants into harmless substances; (*ii*) physical; and chemical treatments which use the physical and/or chemical properties of the contaminants or the contaminated media to destroy, separate or contain the contamination; *and* (*iii*) thermal treatments which use heat to increase the volatility, to burn, decompose, destroy or melt the contaminants.

Biological treatments are generally implemented at lower cost relative to physical, chemical and thermal treatments, and can effectively destroy contaminants leaving little or no residual contamination. Biological treatments take longer time and it is often hard to determine whether contaminants have been completely destroyed. Additionally, microbes may often be sensitive to toxins or highly concentrated contaminants in the soil. Physical and chemical treatments are typically cost effective relative to thermal treatments, and can be completed in shorter time periods relative to biological treatments. Equipments are generally needed, which are widely available. Certain in-situ physical and/or chemical treatment technologies are sensitive to certain soil parameters, which can cause variations in the treatment process performance. Thermal treatments offer the quickest cleanup time but are generally the most costly treatment group. This difference, however, is higher in in-situ than in ex-situ applications. Due to the complex nature of many polluted soils and the fact that pollution, in many situations, is due to the presence of a cocktail of different types of contaminants, it is frequently necessary to apply several remediation techniques (treatment train) to reduce the concentrations of pollutants to acceptable levels.

Table 1 – The remediation technologies for petroleum hydrocarbon contamination in soils used in the DSS (Friend 1996, FRTR 2010, CLU-IN c.2010, EUGRIS c 2010)

TECHNOLOGY	TYPE	TREATMENT	DESCRIPTION
Enhanced Bioremediation (*Syn*. Biostimulation, Bioaugmentation, Enhanced Biodegradation)	In-situ	Biological	Bioremediation uses microorganisms to degrade organic contaminants in soil *in-situ*. The microorganisms break down contaminants by using them as a food source or metabolising them with a food source. Bioremediation can be carried out by either: (*i*) aerobic processes which require an oxygen source, and typically produce carbon dioxide and water as by-products; *or* (*ii*) anaerobic processes in the absence of oxygen, which typically produces methane, hydrogen gas, sulphide, elemental sulphur, and dinitrogen gas as by-products. Enhanced Bioremediation involves providing some combination of oxygen, nutrients, and moisture, and controlling the temperature and pH to enhance bioremediation. Sometimes indigenous microorganisms that have been adapted for degradation of specific contaminants are applied to enhance the process.
Bioventing	In-situ	Biological	Bioventing is a common form of *in-situ* bioremediation that uses extraction or injection wells to circulate air through the ground, which increases oxygen concentrations and stimulate biodegradation. Though some volatilization occurs, the predominant process for contaminant reduction is biodegradation.

Monitored Natural Attenuation (from Natural Attenuation, *syn*. Intrinsic Bioremediation or Bioattenuation)	In-situ	Biological	Natural Attenuation (NA) relies on natural processes to clean up or attenuate pollution in soil and groundwater. NA occurs at most polluted sites. However, the right conditions must exist underground to clean sites properly. If not, cleanup will not be quick enough or complete enough. Monitored Natural Attenuation (MNA) involves the monitoring and/or testing of NA conditions to ensure NA is effective.
Phytoremediation (*Syn*. vegetation-enhanced bioremediation)	In-situ	Biological	Phytoremediation is a process that uses plants to remove, transfer, stabilize, and destroy contaminants in soil and sediment. Contaminants may be either organic or inorganic. The mechanisms of phytoremediation include enhanced rhizosphere biodegradation, phyto-extraction (also called phyto-accumulation), phyto-degradation, and phyto-stabilization.
Air sparging	In-situ	Physical / Chemical	Air Sparging involves the injection of air or oxygen through a contaminated aquifer. Injected air traverses horizontally and vertically in channels through the soil column, creating an underground stripper that removes volatile and semi-volatile organic contaminants by volatilization. The injected air helps to flush the contaminants into the unsaturated zone. Air Sparging is usually is implemented in conjunction with SVE. Oxygen added to the contaminated groundwater and unsaturated zone soils also can enhance biodegradation of contaminants below and above the water table.
Soil Vapour Extraction (*Syn*. In-situ soil venting, In-situ volatilization; Enhanced volatilization)	In-situ	Physical/ Chemical	Soil Vapour Extraction (SVE) is used to remediate unsaturated zone soil. A vacuum is applied to the soil through injection wells to induce the controlled flow of air and remove volatile and some semi-volatile organic contaminants from the soil. SVE usually is performed *in-situ*; however, in some cases, it can be used as an *ex-situ* technology. SVE usually is implemented in conjunction with Air Sparging or Steam Injection
Thermal Treatment (*Syn*. Thermally Enhanced Soil Vapour Extraction)	In-situ	Thermal	Thermal Treatment is a process that uses heat to separate, destroy, or immobilize contaminants. Many different methods and combinations of techniques can be used to apply heat to polluted soil and/or groundwater in situ. The heat can destroy or volatilize organic chemicals. As the chemicals change into gases, their mobility increases, and the gases can be extracted via

			collection wells for capture and cleanup in an ex situ treatment unit.
Soil Washing	Ex-situ	Physical/ Chemical	In Soil Washing, contaminants sorbed onto fine soil particles are separated from bulk soil in a water-based system on the basis of particle size. The wash water may be augmented with a basic leaching agent, surfactant, or chelating agent or by adjustment of pH to help remove organics and heavy metals. Soils and wash water are mixed ex-situ in a tank or other treatment unit. The wash water and various soil fractions are usually separated using gravity settling.
Incineration	Ex-situ	Thermal	Incineration is a heat-based technology that has been used for many years to burn hazardous materials to destroy harmful chemicals. Incineration also reduces the amount of material that must be disposed of in a landfill.
Thermal Desorption	Ex-situ	Thermal	Thermal Desorption is a separation technology that uses thermal treatment to separate contaminants. Pyrolysis and Incineration are used to destroy the contaminants and Vitrification destroys or separates organics and immobilises some inorganics.
Excavation and Disposal (*Syn*. Dig and Dump)	Ex-situ	Other	Excavation and Disposal is one of the earliest remediation methods which involves digging contaminated soil from the location of contamination and dumping it in other locations, mostly landfill sites, where the contaminated soil is not considered a hazard to human health and/or the wider environment. In most cases, the contaminated soil is not treated prior to disposal; however regulatory restrictions waste disposals and steep landfill taxes have brought changes to this practice.

APPENDIX III: DESCRIPTION OF SUSTAINABILITY CRITERIA

The sustainability criteria used in options appraisal of remediation technologies in the DSS are described below in *Table 1* below. These are all from a review of indicators for sustainable remediation by the UK Sustainable Remediation Forum (SURF-UK), who provisionally define sustainable remediation as the practice of demonstrating, in terms of environmental, economic and social indicators, that an acceptable balance exists between the effects of undertaking the remediation activities and the benefits the same activities will deliver (SURF-UK 2009). The criteria fairly evenly distributed across the three elements of sustainable development, the: (*i*) economic; (*ii*) environmental; *and* (*iii*) social elements.

Table 1 – Sustainability criteria used in the Decision Support System (SURF-UK 2009)

CRITERIA	DESCRIPTION
ECONOMIC element of sustainability	
Direct costs	Costs represent the use of economic resources. Direct costs are those that effectively affect the "bottom line" of the organisation or organisations that would undertake the project being considered. As for environmental resource utilisation, the usual desire is to minimise economic resource utilisation (so that economic resources can generally be applied most effectively – particularly important for a public administration).
Indirect costs	Costs may also be indirect, and these indirect costs may not accrue to the project or organisations undertaking it, for example the long term impact of reducing investment to deal with an overly expensive project, or costs needed for supporting infrastructure measures. Direct and indirect (or consequential) costs have been considered separately to take into account that they may affect different groups, and that they are estimated differently.
Time span	Initiatives and projects with a short life span represent a poorer investment as the duration of the services they provide is limited. Project risks include

	issues such as the reliability of projects / technologies; technology status and maturity, issues of due diligence and taking decisions that affect the susceptibility of an activity to environmental hazards (such as flooding).
ENVIRONMENTAL element of sustainability	
Impacts on other resources (air, water, soil)	Impacts on air include issues of air quality (pollution) – e.g. NOx, SOx, acid, particulates, VOC; climate change, e.g. CO_2, N_2O; and ozone depleting substances (see also social – human health). Impacts on water include: emissions of nutrients (especially N and P), particulates (sediment inputs), impacts on pH and redox, emissions of other dissolved contaminants, transfer of pathogens, impacts if flow rates are low. Impacts on soil include: changes in biological functions, chemical functions and physical functions, accumulation of contamination, biological "contamination", physical contamination. Includes geotechnical performance (e.g. subsidence risks)
Impacts on ecological systems	An over-riding concern for environmental impacts is their consequences for ecology, both in terms of biodiversity (from a conservation perspective) and from the perspective of providing services necessary for the sustenance of life. Includes risks / impacts on ecological functioning and biodiversity, including imported species such as weeds
Intrusiveness	Environmental impacts may not be readily tangible, in many cases impacts may arise from noise, light or simply a visual impact. Includes impacts on the built environment, conservation issues (e.g. preservation of archeologically important strata), impacts on landscape. Includes also impacts from flooding, risks from flooding and avoidance of flooding risks.
Resource use and waste by-products	Resource utilisation is an important consideration in sustainability appraisal. Environmental resources considered typically include materials and energy, and are important both in terms of their depletion, and also the environmental impacts of their production. Other resources are also important, for example water use, land use, use of landfill capacity and other downstream waste management capacity, and also the built environment and archaeological remains which may be altered or destroyed. Waste by-products are also part of the resource cycle. Utilisation of non-renewable resources tends of course to be more significant than the use of renewable resources – depending on the environmental costs of the resource production. Includes waste minimisation
SOCIAL element of sustainability	
Human health	Achieving satisfactory long-term risk management, dealing with issues of risk perception, effects of noise, odour, dust and bio-aerosols; needs to consider acute versus chronic risks, occupational exposure and health and safety issues of workers on site.
Impacts on neighbouring land	Projects can cause aggravation for example by removing or reducing public access to land, by increasing traffic and congestion, by closing access routes;

	or more generally by being insensitive to site neighbours. Other sources of aggravation may be nuisance issues such as noise, light pollution, smells, litter and debris off site. This category also includes traffic issues at all scales, and issues related to crime, disorder and public safety
Uncertainty, evidence and fit with policy	Sustainable development policy in the UK is explicitly described as evidence based. Consequently it is important to consider in the sustainability appraisal the quality of the evidence presented in support of claims for the proposed options being considered. A related issue is uncertainty. The lower the level of uncertainty over possible outcomes for an option being considered, the more likely that option would be successful if implemented in practice. Another important consideration is how assertions of sustainability can be verified once a project development is underway or has been implemented, and operations have begun. In broad terms this category considers the quality of information going into the sustainability appraisal. Issues of policy include meeting all clean-up and planning requirements.

APPENDIX IV: DEVELOPEMENT OF THE DATABASE COMPONENT

The Decision Support System (DSS) database was developed as a Relational Database (RDB), using the MySQL Relational Database Management System (RDBMS) database server. A RDB has been used to overcome the limitations of static *flat-file* data storage systems. In the flat-file system data is stored in text files, where the data is usually written to be executed by specific programs. Additionally, in the flat-file storage system, the database is distributed across a number of files, which are executed line by line until a result is found. Therefore flat file databases do not exist as a single integrated structure. In RDBs the database is a central resource that is designed and managed in its own right (Eaglestone 1991). This approach to data operation and management is in line with the component-based software development approach used for developing the DSS, which will allow parts of the database to evolve without affecting the overall database. And because RDB models are standardised they are independent of RDBMSs, and can be therefore be implemented on other RDBMS *e.g*. Microsoft SQL Server, PostgreSQL, Oracle *etc*. The DSS database was designed as a RDB model, using principles of RDB designed (*below*). The database was then created based on the designed and optimised RDB model, and implemented in MySQL RDBMS. The DSS accesses the database via its *User Interface*.

DATABASE ENTITIES

RDBs store data in relational tables called *entities*, which are searchable with a RDBMS. The properties of each entity in the RDB are known as its *attributes*, which are represented by the columns of the table. Data is stored in rows in RDB tables, with each row representing an independent record. Each row consists of a set of individual values corresponding to the attributes. The data for the DSS was stored as relational tables; with each row representing an independent record, and each column representing its attribute, each of which as a unique name, data type and size. The values of each record represent facts corresponding to each attribute. The entities for the DSS were identified from publicly available DSS and Decision Support Tools (DSTs) for contaminated land management, particularly petroleum hydrocarbon contamination; technical reports and guidance; and legislative requirements. The identified entities are presented in *Table 1* below.

Table 1 – Entities in the database model

ENTITY	NOTATION	DESCRIPTION
Site	SITE	For storing site details
Preliminary qualitative assessment	PRELIM_QRA	For storing site details for preliminary risk assessment
Site samples	SAMPLE	For storing details of collected site samples
Environment Agency (EA) Soil Guideline Values (SGVs)	GAC_EA	Contains EA-based generic assessment criteria for stored contaminant in the database.
Generic quantitative risk assessment using EA SGV	GQRA_EA	For storing results of chemical analysis of measured site sample concentrations to be used for comparing with generic assessment criteria for risk assessment
Land use types	LAND_USE	Contains land use types. These are used for defining EA-based SGVs for different land-use scenarios.
Dutch Intervention Values (DIV) Generic Assessment Criteria (GAC)	GAC_DIV	Contains DIV-based generic assessment criteria for stored contaminant in the database.
Generic quantitative risk	GQRA_DIV	For storing results of chemical analysis of

assessment using DIV		measured site sample concentrations to be used for comparing with generic assessment criteria for risk assessment
Contaminated media	CONTAMINATED_MEDIA	Contains the type contaminated media to be assessed using DIVs.
Contaminants	CONTAMINANT	Contains details of all contaminants to be used for decision support, including indicator compounds and hydrocarbon fractions.
Contaminant type	CONTAMINANT_TYPE	Contains the different contaminant types.

RELATIONSHIPS BETWEEN ENTITIES

A relation represents an association between two entities. The relationships between the entities in the DSS are illustrated in the *Entity Relationship Diagram* (ERD) in *Figure 1*. Entities are represented by boxes, and the relationships are represented by the lines between entities. Entities that do not depend on other entities are known as *independent entities*, and are represented by single-bordered boxes, and the entities which cannot exist without a parent entity, are known as *dependent entities* and are represented by the double-bordered boxes in the ERD. Relationships have different types of cardinality: one-to-one (1:1); one-to-many (1:M); *and* many-to-many (M:M). The cardinalities of all the relationships are represented in the ERD using the *Crow's Foot Notations*. The many-to-many relationships between the `SAMPLE` and `GAC` and between `REM_TECH` and `SUST_CRI` tables are solved using *associative entities* (`OP_APP` and `GQRA` tables, respectively), which store additional data that does not fit into the attribute list of either entity in the M:M relationships.

ATTRIBUTES OF ENTITIES

An attribute is a unit of fact that describe the properties of an entity, which is represented by the columns of a table. Each row (record) of an entity has a value for each of its attributes (which could be null) with each row having the same data type and size for the same attribute. The attributes of each of the entities are presented in *Table 2*. These are based on the input data that are required for each stage of the contaminated land management process, particularly the management of petroleum hydrocarbon contaminants. These were identified from contaminated land DSS and technical reports and guidance. A description of all the attributes for all the entities identified for is given in *Table 3* below.

Table 2 – Entities and their corresponding attributes

ENTITY	ATTRIBUTES
SITE	id, site_name, site_loc, site_description, authors
PRELIM_QRA	site_id, c_type, c_luse, l_euse, nnluse, snluse, wluse, enluse, perm, vuln, gwater, swater, abs_lic
SAMPLE	id, site_id, sample_name, sample_media, sample_description
GAC_EA	id, contaminant, land_use, gac_value
GQRA_EA	id, sample_name, contaminant, gac_value, ms_conc0, ... , ms_conc34
LAND_USE	id, land_use_type
GAC_DIV	id, contaminant, media, div_value
GQRA_DIV	id, sample_name, contaminant, div_value, ms_conc0, ... , ms_conc34
CONTAMINATED_MEDIA	id, media
CONTAMINANT	id, name, type_id
CONTAMINANT_TYPE	id, type

Table 3 – Description of attributes of all the entities

ATTRIBUTE	NOTATION	DESCRIPTION
id	id	Each record of each entity in the database has a unique identifier
Site name	site_name	Name of the site being manages
Site location	site_location	Location of the site
Site description	site_description	Site description is used for generating report
Report author(*s*)	authors	Name of report author(*s*) is used for generating report
Contamination type	c_type	The type of contamination being managed
Current land use	c_luse	Current land use is used for selecting the appropriate generic assessment criteria
Land end use	l_euse	Land end use is used for selecting the appropriate generic assessment criteria
Northern neighbouring land use	nnl_use	Northern neighbouring land use on all for selecting the appropriate generic assessment criteria
Western neighbouring land use	wnl_use	Western neighbouring land use on all for selecting the appropriate generic assessment criteria
Southern neighbouring land use	snl_use	Southern neighbouring land use on all for selecting the appropriate generic assessment criteria
Eastern neighbouring land use	enl_use	Eastern neighbouring land use on all for selecting the appropriate generic assessment criteria
Abstraction license	abs_lic	Abstraction license is used for selecting the appropriate generic assessment criteria
Soil vulnerability	dis_lic	Site vulnerability
Aquifer permeability	pol_inc	Aquifer permeability
Groundwater	gwater	Groundwater details are used for selecting the appropriate generic assessment criteria
Surface water	swater	Surface water details are used for selecting the appropriate generic assessment criteria
Sample name	sample_name	Sample name is used for identifying collected site samples
Sample type	sample_type	Sample type is used for identifying the type/source of sample

Sample description	sample_description	Sample description is used for generating report
Measured sample concentration	ms_conc0[] ... ms_conc4[]	Measured sample concentration is the result of chemical analysis of collected site samples
EA based generic assessment criteria value	gac_ea	Generic assessment criteria (GAC) are used for generic quantitative risk assessment. The derivation of GAC is covered in Appendix C
Dutch intervention generic assessment criteria source	gac_div	The source of the GAC, which could be the EA, LQM/CIEH, DIV or the ICRCL. The details of GAC sources is covered in Appendix C
Land use type	land_use_type	For selecting appropriate GAC values based on land end use
Contaminant name	cont_name	Contaminant name is used for identifying the different contaminants used in the database. The details of contaminants used in the database is covered in Appendix C
Contaminant type	type	Type of contaminant
Technology name	tech_name	Technology name is used for identifying remediation technologies. A description of all remediation technologies used is covered in Appendix D
Contamination zone	cont_zone	Contamination extent is used for defining the extent of contamination present at a site. This attribute is used for remediation design.
Resource use	resU	Equipment use is used for selecting the level of equipment use for remediation. This attribute is used for remediation design.
Energy use	energy	Energy use is used for selecting the level of energy use for remediation. This attribute is used for remediation design.
Waste by-products	waste	The level of waste produced by a remediation technology. This attribute is used for remediation design.
Transport	transport	If transport is needed for equipment, materials or waste b-products. This attribute is used for remediation design.
Cost	relC	The cost of the remediation technology relative to the other technologies in the database. This attribute is used for remediation design.
Time	relT	The relative time of it takes for clean-up. This attribute is used for remediation design.

Treatment train	treat	Treatment train is used for selecting remediation technologies that are part (or not) of a treatment train. This attribute is used for remediation design.

ASSIGNING KEYS

In RDBs, unique identifiers known as *keys* are used for enforcing database integrity. Primary Keys (PK) are used for uniquely identifying each record in an entity. In some cases, multiple attributes known as *composite keys* are needed to uniquely identify records. A Foreign Key (FK) is an attribute of an entity that is used as the PK of another entity. The FK and the referenced PK must have the same data types, and ideally size. FKs are used to ensure referential integrity across the database is maintained. Referential integrity ensures all data cross-referenced from within the database is also described within the database, which reduces data duplication and redundancy (Eaglestone 1991). Indexing both FK and referenced PK enables the quick checking of data integrity, optimising database performance. Each entity in the database has been assigned a PK to uniquely identify each record. Weak entities have been assigned FKs, which are the PKs of the entities they depend on. The ERD diagram in *Figure 1* shows all attributes of the entities, the data types of the attributes, and the respective keys of each entity.

DATA TYPES OF ATTRIBUTES

Each record has a data type specified by its attributes. Data types determine which type of information will be stored for each attribute, its value, and its size. Giving the correct data type and size optimises the database performance. Different numeric, date/time and string data types have been used for the attributes, with varying sizes. All the data types of the attributes in the DSS and their respective values are presented in *Table 4*.

Table 4 - Data types and sizes of attributes

ATTRIBUTE	DATA TYPE	ATTRIBUTE	DATA TYPE
Id	INT (11)	sample_media	VARCHAR (14)
site_name	VARCHAR (50)	sample_descr	VARCHAR (255)
site_loc	VARCHAR (50)	contaminant_name	VARCHAR (50)
site_descr	VARCHAR(255)	contaminant_type	VARCHAR (!5)
authors	VARCHAR (100)	contaminated_media	VARCHAR (11)
c_type	VARCHAR (22)	div_value	DECIMAL (7, 2)
c_luse	VARCHAR (25)	gac_value	DECIMAL (7, 2)
l_euse	VARCHAR (25)	land_use_type	VARCHAR (25)
nnl_use	VARCHAR (25)	ms_conc0[] – m_conc4[]	DECIMAL (7,2)
wnl_use	VARCHAR (25)	technology	VARCHAR (50)
snl_use	VARCHAR (25)	Zone	VARCHAR (6)
enl_use	VARCHAR (25)	relC	VARCHAR (7)
perm	VARCHAR (15)	relT	VARCHAR (5)
vuln	VARCHAR (15)	techT	VARCHAR (3)
gwater	VARCHAR (3)	Treat	VARCHAR (3)
swater	VARCHAR (3)	resU	VARCHAR (7)
abs_lic	VARCHAR (3)	Waste	VARCHAR (7)
sample_name	VARCHAR (10)	Trans	VARCHAR (3)

Table 5 - Description of SQL data types used

DATA TYPE	RANGE	DESCRIPTION
INT ( )	-2147483648 to 2147483647 / 0 to 4294967295 (Unsigned)	INT data type is used to store exact numeric data values
BLOB ( )	64KB	BLOB is a binary large object that can hold a variable amount of data to a maximum of 64KB
BOOL ( )	0, 1	Short for BOOLEAN which means that each column may have one of a specified possible values.
DATE	1000-01-01 9999-12-31	DATE data type for representing temporal values
DECIMAL(,)	Varies	DECIMAL data type is used to store exact numeric data values.
TEXT ( )	64KB	TEXT values are treated as non-binary (character) strings
VARCHAR ( )	1 – 255	VARCHAR is a variable-length string, which can be specified as a value from 0 to 255

NORMALISATION

Database normalisation is a process of optimising an RDB model by refining and organising data to ensure all data dependencies are logical. Normalising databases reduces data redundancy and operational anomalies, and improves the overall efficiency and performance of the database. Database normalisation modifies the RDB model using a series of progressive restrictions, each of which excludes certain undesirable properties from the database design (Eaglestone 1991). The restrictions are called *Normal Forms* (NFs), and normalisation ensures the RDB model that does not violate the NFs. Many NFs have been defined, of which there are six established in database theory (Eaglestone 1991):

- The *First Normal Form* (1NF) which is concerned with simplifying structures in the database to ensure each attribute has only single values

- The *Second Normal Form* (2NF), *Third Normal Form* (3NF) and the *Boyce-Codd Normal Form* (BCNF, also 3.5NF) which are concerned with eliminating duplication of data that represent single value facts
- The *Fourth Normal Form* (4NF) and *Fifth Normal Form* (5NF) which are concerned with eliminating the duplication of data that represent multi-valued facts.

In most practical applications, RDB models are only normalised to 3NF. This is because although complete normalisation is desirable, it can introduce complexity in application. There is also a trade-off between complete normalisation and database performance. The more progressively normalised an RDB model is, the more tables it will contain, which results in more SQL operations, potentially leading to decrease in performance. To that effect, the DSS RDB model has been normalised to 3NF. Although a higher level of normalisation cannot be achieved without satisfying previous level(*s*), normalised tables can be created directly without iterating through the lower forms.

All the tables in the ERD have been normalised to 3NF. This is because all the tables have a PK, and contain no repeating values within any column of all the tables. This satisfies 1NF requirements. 2NF requires all *non-key* columns to be dependent on the entire PK of the table, and for composite PKs, all *non-key* columns must depend on the whole not part of the PK. All the tables are in 2NF because all the *non-key* attributes of each of the tables in the RDB model are fully dependent on the respective table's PK. 3NF requires that all columns in a table directly depend on the PK of that table and not on other attributes, which all tables in the model satisfy (*Fig. 1*).

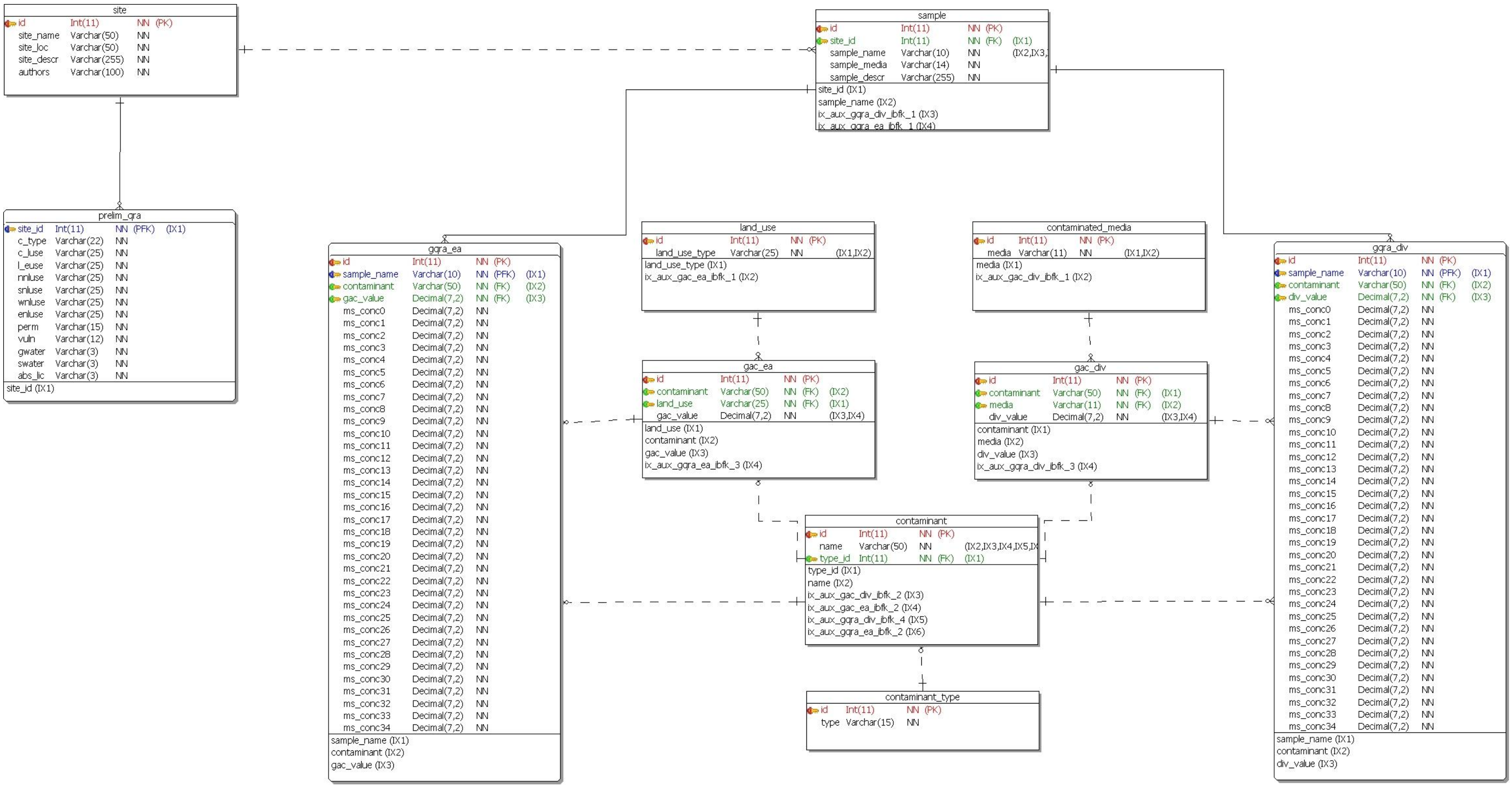

Figure 1 – The normalised ERD for the RDB data model showing entities with their corresponding attributes, data types, and keys

DATABASE IMPLEMENTATION

The DSS database was developed from the RDB model develeoped above using the international standard database language `SQL`, embedded in MySQL RDBMS on a Linux operating system. The database was created using the MySQL `CREATE DATABASE` command and the `USE` command to select the created database for use. All MySQL commands are terminated by a semicolon. In creating the DSS database, all SQL commands are in uppercase, and database and table names are in lowercase:

```
CREATE DATABASE database_name;
USE database_name;
```

The tables in the database were also created using the `CREATE` command (*below*) with rows separated by comma, where `field` indicates the attribute name; `data` indicates the associated data type of the attribute; `null` indicates whether the record has null values or not; `key` indicates whether the column is indexed as PK or FK; `default` indicates the default value assigned to the attribute; *and* `extra` indicates additional information about the column. An `auto_increment` value is used for columns with the `AUTO_INCREMENT` attribute or empty otherwise. The AUTO_INCREMENT attribute is used for automatically generating a unique identity for new records (MySQL *c*2010).

```
CREATE TABLE table_name (
      field_1 data null key default extra,
      field_2 data null key default extra,
      field_3 data null key default extra
         ) ENGINE=STORAGE_ENGINE;
```

FKs have been used to establish relationships between the tables in the database. However, MySQL only allows the use of FKs with the `InnoDB` storage engine, and therefore `InnoDB` storage engine was used in developing the DSS database. The storage engine is the component of the RDBMS that is used for creating, reading, updating and deleting data from the database. The referenced table must also be in `InnoDB`, must have an index and a PK and have the data type of the FK must be the same as that of the referenced PK so that they can be compared without a type conversion. Ideally the size of the data type should also be the same. InnoDB requires indexing FKs and the referencing the corresponding PKs so that FK checks can be fast and not require a complete table scan. For example, a `PARENT` and `CHILD` FK relationship can be written using the syntax (MySQL *c*2010):

```
CREATE TABLE parent (
      id INT NOT NULL PRIMARY KEY
         ) ENGINE=INNODB;

CREATE TABLE child (
      id INT,
      parent_id INT,
      INDEX par_ind (parent_id),
      FOREIGN KEY (parent_id)
      REFERENCES parent (id)
      ON DELETE CASCADE
         ) ENGINE=INNODB;
```

This creates a parent and child table with a many-to-many relationship where many parents can have many children. `InnoDB` rejects any `INSERT` or `UPDATE` operation that attempts to create an FK value in a child table if there is no matching candidate key value in the parent table. The action `InnoDB` takes for any `UPDATE` or `DELETE` operation that attempts to update or delete a candidate key value in the parent table that has some matching rows in the child table is dependent on the referential action specified using `ON UPDATE` and `ON DELETE`. `InnoDB` supports different options regarding the action to be taken when attempt is made to delete or update a row from a parent table with one or more matching rows in the child table, The database of the DSS, `wiDSS` (which stands for *Web-based Intelligent Decision Support System*), was created and selected for use using:

```
CREATE DATABASE widss;
USE widss;
```

The schema defining the tables their attributes, data types and sizes of all the tables in the database are given below, where `FIELD` represents the attribute name; `TYPE` represents the data type and size; `NULL` and `DEFAULT` indicates whether the attribute has a null value or not; `KEY` indicates whether the column is indexed, and the type of index the column has; and `EXTRA` indicates whether a record has an `auto_increment` value. If the `KEY` field is empty, the column is not indexed. The `KEY` values used are `PRI`, for PKs; `UNI` for unique keys; *and* `MUL`, which indicates multiple occurrences of a given value are permitted within the column. If more than one of the Key values applies to a given column of a table, `KEY` displays the one with the highest priority, in the order `PRI`, `UNI`, `MUL` (MySQL *c*2010).

Table 6 – Schema for SITE table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
site_name	varchat (50)	NO		NULL	
site_location	blob	NO		NULL	

site_description	varchar (50)	NO		NULL	
authors	varchar (50)	NO		NULL	

Table 7 – Schema for PRELIM_QRA table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
site_id	int (11)	`NO	MUL	NULL	
c_type	varchar (12)	NO		NULL	
c_luse	varchar (12)	NO		NULL	
l_euse	varchar (12)	NO		NULL	
nnl_use	varchar (12)	NO		NULL	
wnl_use	varchar (12)	NO		NULL	
snl_use	varchar (12)	NO		NULL	
enl_use	varchar (12)	NO		NULL	
Perm	varchar (12)	NO		NULL	
Vuln	varchar (12)	NO		NULL	
Gwater	varchar(3)	NO		NULL	
Swater	varchar(3)	NO		NULL	
abs_lic	varchar(3)	NO		NULL	

Table 8 – Schema for SAMPLE table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
site_id	int (11)	NO	MUL	NULL	
sample_name	varchar (10)	NO		NULL	
sample_media	varchar (14)	NO		NULL	

sample_description	varchar (255)	NO		NULL	

Table 9 – Schema for CONTAMINANT table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
cont_name	varchar (30)	NO		NULL	
type_id	int(11)				

Table 10 – Schema for CONTAMINANT_TYPE table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
Type	varchar(15)				

Table 11 – Schema for CONTAMINATED_MEDIA table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
Media	varchar (11)	NO		NULL	

Table 12 – Schema for LAND_USE table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
land_use_type	varchar (25)	NO		NULL	

Table 13 – Schema for GAC_EA table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
contaminant	int (11)	NO	MUL	NULL	
gac_value	decimal (7, 2)	NO		NULL	
land_use	varcahr (25)	NO		NULL	

Table 14 – Schema for GAC_ DIV table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
contaminant	int (11)	NO	MUL	NULL	
Media	varchar(11)				
gac_value	decimal (10, 4)	NO		NULL	

Table 15 – Schema for GQRA_EA table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
id	int (11)	NO	PRI	NULL	auto_increment
site_id	int (11)	YES	MUL	NULL	
contaminant	varchar (50)	YES	MUL	NULL	
gac_value	decimal(7,2)	YES	MUL	NULL	
ms_conc0	decimal(7,2)	YES	MUL	NULL	
ms_conc1	decimal(7,2)	YES	MUL	NULL	
ms_conc2	decimal(7,2)	YES	MUL	NULL	
ms_conc3	decimal(7,2)	YES	MUL	NULL	
ms_conc4	decimal(7,2)	YES	MUL	NULL	

Table 16 – Schema for GQRA_DIV table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
id	int (11)	NO	PRI	NULL	auto_increment
site_id	int (11)	YES	MUL	NULL	
Contaminant	varchar (50)	YES	MUL	NULL	
div_value	decimal(7,2)	YES	MUL	NULL	
ms_conc0	decimal(7,2)	YES	MUL	NULL	
ms_conc1	decimal(7,2)	YES	MUL	NULL	
ms_conc2	decimal(7,2)	YES	MUL	NULL	
ms_conc3	decimal(7,2)	YES	MUL	NULL	
ms_conc4	decimal(7,2)	YES	MUL	NULL	

Table 17 – Schema for GAC_TYPE table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
Type	varchar (3)	NO		NULL	

Table 18 – Schema for TECHNOLOGY table

FIELD	TYPE	NULL	KEY	DEFAULT	EXTRA
Id	int (11)	NO	PRI	NULL	auto_increment
technology	varchar (40)	NO		NULL	
Zone	varchar (6)	NO		NULL	
relC	varchar (7)	NO		NULL	
relT	varchar (5)	NO		NULL	
techT	varchar(3)	NO		NULL	
Treat	varchar(3)	NO		NULL	

resU	varchar (7)	NO	NULL
Waste	varchar(7)	NO	NULL
Treans	varchar (3)	NO	NULL

APPENDIX V: DEVELOPMENT OF THE DECISION MODEL

The decision model used in the Decision Support System (DSS) was developed using the Analytical Hierarchy Process (AHP). The AHP is a structured Multi Criteria Decision Analysis (MCDA) method that is used to systematically compare decision outcomes. The AHP process broadly consists of four key stages: (*i*) problem formulation; (*ii*) weights valuation; (*iii*) weights aggregation; *and* (*iv*) sensitivity analysis. In developing the decision model, the decision goal, alternatives and criteria were first identified (*Table 1*). The goal of the decision model is the selection of the most sustainable remediation technology for the cleanup of contaminated land, given site specific parameters. The sustainability criteria and alternatives were derived from the published literature, guidelines, technical reports and expert judgement. All the alternatives used are established remediation technologies that are suitable for cleaning petroleum hydrocarbon contamination. The sustainability criteria used are based on the indictors for sustainable remediation identified by the Sustainable Remediation Forum UK. A description of all the remediation technologies the criteria is provided in Appendix II and III respectively Appendix III.

The decision goal, criteria, sub-criteria and alternatives were decomposed into a four level hierarchical structure (*Figure 1*). A four level hierarchy was developed because it has been observed that criteria with a large number of sub-criteria tend to receive more weight than when they are less detailed, it is recommended that for hierarchies with large numbers of elements, the elements should be arranged in clusters so they do not differ in extreme ways (Ishizaka and Ashraf 2009). The hierarchy provides an overall view of the relationships within the different elements of the decision problem and allows for the comparison elements of the same order of magnitude with respect of the overall goal (Saaty 1987).

Table 1 – The sustainability criteria, sub-criteria and alternatives used

CRITERIA	SUB-CRITERIA	ALTERNATIVES
Economic	Direct costs (EC1)	Bioventing (A1)
	Indirect costs (EC2)	Enhanced bioremediation (A2)
	Time span (EC3)	Monitored natural attenuation (A3)
Environmental	Impacts on other resources (EN 1)	Phytoremediation (A4)
	Impacts on ecological system (EN2)	Air sparging (A5)
	Intrusiveness (EN3)	Soil vapour extraction (A6)
	Resource use and waste by-products (EN4)	Thermal treatment (A7)
Social	Impacts on human health (S1)	Soil washing (A8)
	Impacts on neighbouring land (S2)	Incineration (A9)
	Uncertainty, evidence and policy (S3)	Thermal desorption (A10)
		Excavation and disposal (A11)

Table 2 – Saaty's fundamental 9-point scale for pairwise comparisons

Intensity of importance	DESCRIPTION
1	Criterion i and j are of equal importance
3	Criterion i is moderate more important than criterion j
5	Criterion i is strongly more important than criterion j
7	Criterion i is very strongly more important than criterion j
9	Criterion i is extreme more important than criterion j
2, 4, 6, 8	For compromise between above values

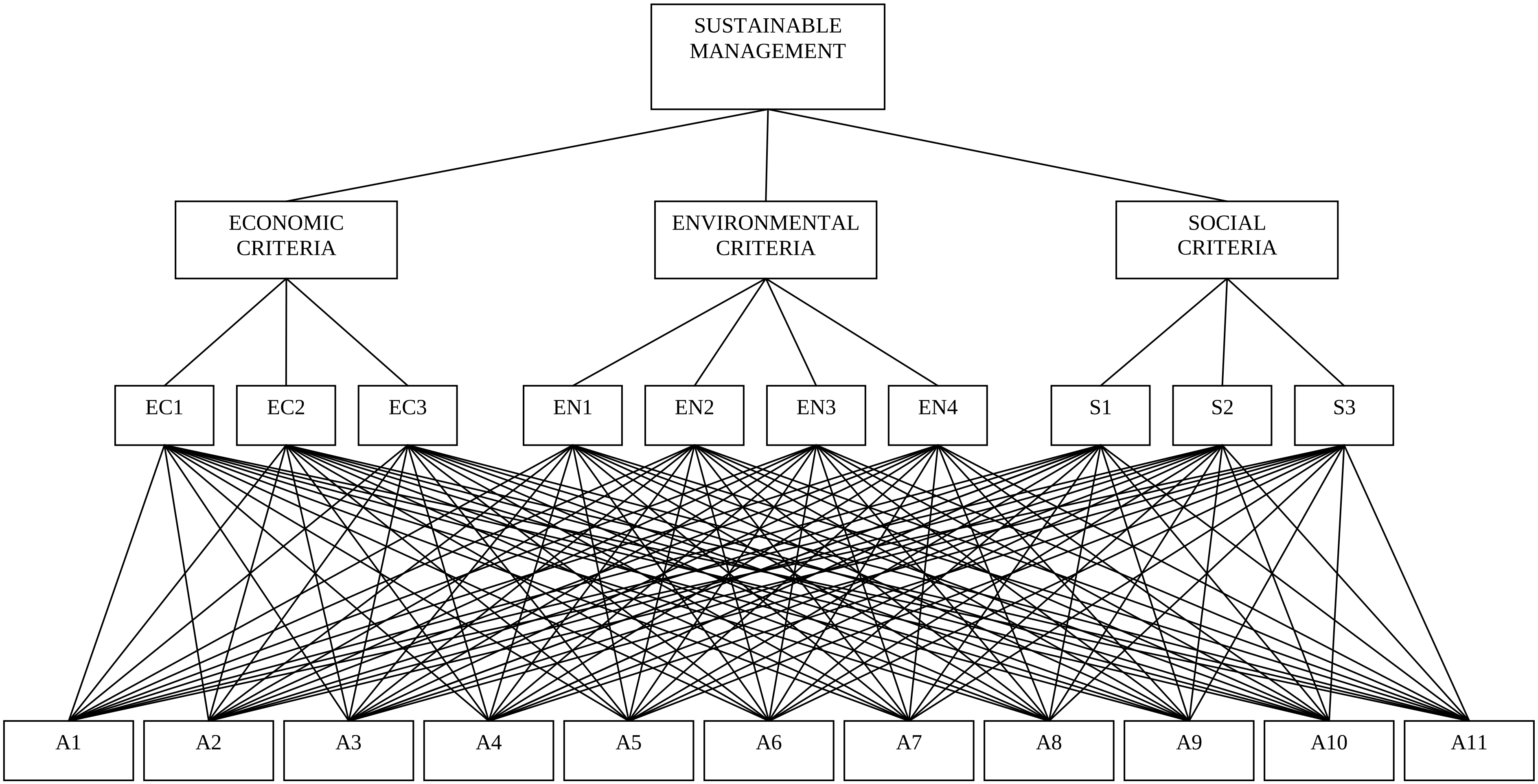

Figure 1 – The decision hierarchy for the selection of the most sustainable remediation technology

Both qualitative and quantitative information were used for pairwise comparisons at each hierarchical level. The pairwise comparisons were carried out using the Saaty fundamental scale of absolute numbers (*Table 2*), which is used to assign numerical values to both quantitative and qualitative judgements by asking questions like 'with respect to criterion ***x***, how much more important or dominant is alternative *i* to *j*?' The elements of each hierarchical level were prioritised based on their relative importance to every other element in their hierarchy, with respect to a parent element, *i.e*. criteria were compared with respect to goal, sub-criteria to each of their parent criteria, and alternatives with respect to each sub-criterion. The results of the pairwise comparisons were recorded in positive consistency matrices, where the overall priorities of at each level of the hierarchy = 1.0.

The decision model contains a total of 14 pairwise comparison matrices, consisting of a total of 565 pairwise comparisons. After the pairwise comparison matrices were completed, priorities were derived using the Eigen value method, by normalising each column of each matrix, to derive the normalised principal Eigen (priority) vector. After that, the consistency of the comparison matrix was calculated. The local priorities across all the criteria are aggregated and normalised to derive their overall priorities. In the last stage of the AHP process, sensitivity analysis was carried out. Sensitivity analysis involves slightly modifying the weights of the criteria to observe the impact on the priority weights. The results are said to be robust if the ranking does not change in sensitivity analysis.

The derivation of priorities is demonstrated with the first comparison matrix, the pairwise comparisons of sustainability criteria with respect to the goal (*Table 3*). A value one 1 is recorded when an alternative is compared to itself in the comparison matrix, such that: $a_{ii} = 1$. Only the upper triangular matrix needs to be filled in – the bottom triangular matrix is the reciprocal value of the upper matrix such that for all $a_{ij} = k$, the corresponding reciprocal value is $a_{ji} = \frac{1}{k}$. All the elements of the completed matrix are positive and reciprocal such that $a_{ij} > 0$. The number of judgments J that were made in each comparison matrix was $J = \frac{n(n-1)}{2}$ = 3, where n is the size of the matrix (Saaty 1990). This is because only the upper triangular matrix needs to be filled in – the bottom triangular matrix is derived as the reciprocal value of the upper matrix such that for all $a_{ij} = k$, the corresponding diagonal reciprocal value is $a_{ji} = \frac{1}{k}$.

Table 3 – Pairwise comparisons of sustainability criteria with respect to goal

	Eco.	Env.	Soc.
Eco.	1.0	$\frac{1}{3}$	$\frac{1}{5}$
Env.	3.0	1.0	½
Soc.	5.0	2.0	1.0

Table 4 – The relative weights of the pairwise comparisons

	Eco.	Env.	Soc.
Eco.	1.0	$\frac{1}{3}$	$\frac{1}{5}$
Env.	3.0	1.0	½
Soc.	5.0	2.0	1.0
∑ COL	9	$\frac{10}{3}$	$\frac{17}{10}$

Table 5 – The normalised relative weights of the pairwise comparisons

	Eco.	Env.	Soc.
Eco.	$\frac{1}{9}$	$\frac{1}{10}$	$\frac{2}{17}$
Env.	$\frac{3}{9}$	$\frac{3}{10}$	$\frac{5}{17}$
Soc.	$\frac{5}{9}$	$\frac{3}{5}$	$\frac{10}{17}$
Norm. COL	1.0	1.0	1.0

All the elements of the completed matrix are positive and reciprocal such that $a_{ij} > 0$. The normalised relative weights were calculated by adding the values of each column of the reciprocal matrix (*Table 4*), and then dividing each value of the column by the sum, which = 1.0 (*Table 5*). The priority vector, the Eigen vector, was computed by averaging each row of the comparison matrix using $A \times w = \lambda_{\max} \times w$, where *A* is the comparison matrix; *w* is the normalised principal Eigen vector; *and* $\lambda_{\max}$ the priority value of *A* (Saaty 1987).

Table 6 – The normalised principal Eigen (priority) vector

	Eco.	Env.	Soc.		∑ ROW			Eigen vector
Eco.	$\frac{1}{9}$	$\frac{1}{10}$	$\frac{2}{17}$		0.329			0.10959
Env.	$\frac{3}{9}$	$\frac{3}{10}$	$\frac{5}{17}$	=	0.927	*	$\frac{1}{3}$	0.30915
Soc.	$\frac{5}{9}$	$\frac{3}{5}$	$\frac{10}{17}$		1.744			0.58126
∑ COL	1.0	1.0	1.0					∑ = 1.0

The normalised priority vector *w* was obtained by averaging across the rows (*Table 6*). The sum of all the elements of the priority vector = 1.0. The priority vector represents the relative weights of the criteria with respect to the overall goal, which for the comparison matrix is 58%, 31% and 11% for the social, environmental and economic criteria respectively. In most cases, the sustainability criteria are given equal weights, however in this case the sustainability criteria have different weights representing the influence of each criterion to the decision problem, which rates the protection of human health (a social sub-criterion) above all other criteria in the decision hierarchy.

Finally, the consistency of the comparison matrix was calculated. Although the AHP allows for inconsistency in decision-making, the AHP provides a method of calculating the decision maker(*s*) inconsistency, the Consistency Index (*CI*) which is used to determine the degree of consistency in a comparison matrix. A threshold value of ≤ 0.10 is deemed acceptable. A larger CI value will disrupt consistent measurement, and lower CI value would make an insignificant change in measurement (Saaty 2004, 1990). Other methods have been developed for deriving such priorities in an effort to reduce rank reversal. The most common of which is the geometric mean (*also* logarithmic least squares) method (Ishizaka 2004, Ishizaka and Labib 2009). It has been mathematically demonstrated that the Eigen vector solution is the best approach (Saaty 1990).

A comparison matrix is consistent if for all i, j, k the raking is transitive, such that: $a_{ij} * a_{jk} = a_{jk}$. In consistent reciprocal matrices, the principal Eigen (priority) value $\lambda_{\max}$ should be equal to the size of the comparison matrix *n*, such that: $\lambda_{\max} = n$. The principal Eigen value is calculated by multiplying the Eigen vector with the sum of the criteria weights of each of its reciprocal matrix and then adding all the products:

$$\lambda_{\max} = 9(0.10959) + \frac{10}{3}\left(0.30915\right) + \frac{17}{10}\left(0.58126\right) = 3.004952$$

The CI is calculated as $CI = \frac{\lambda_{\max} - n}{n-1}$, where $\lambda_{\max}$ is the principal Eigen value and n is the dimension of the comparison matrix. The CI of the comparison matrix was calculated:

$$CI = \frac{3.004952 - 3}{3 - 1} = 0.002476$$

The *Consistency Ratio* (CR), which is the ratio between CI and RI, the *Ratio Index*, was then calculated using: $CR = \frac{CI}{RI}$ (*Table 7*). The RI is the average CI of 500 randomly filled matrices (Saaty 1977). Other RI values have been calculated by other researchers, and alternative methods exist for measuring consistency (Ishizaka and Labib 2009).

Table 7 – Random index values (Saaty 1977)

n	1	2	3	4	5	6	7	8	9	10
RI	0.00	0.00	0.58	0.90	1.12	1.24	1.32	1.41	1.45	1.49

$$CR = \frac{0.002476}{0.58} = 0.00427$$

The $CR = 0.00427$ < 0.1 %, therefore the comparison matrix is considered consistent. This result was validated the Expert Choice 11.5™ (EC) AHP software (EC 2009). The remaining 13 comparison matrices were calculated the same way and validated using EC, with number of pairwise comparisons and the level of difficulty increasing as the size of the matrix increased. The comparison matrices and their CI are presented in *Tables 8 to 21*. The relative values of each criterion for each of the alternatives were derived from the literature and technical reports (*Table 11*) for the pairwise comparisons of the alternatives.

Table 8 – Pairwise comparisons of economic sub-criteria with respect to economic criteria

	EC1	EC2	EC3
EC1	1.0	1.0	1.0
EC2		1.0	1.0
EC3			1.0
Inconsistency 0.00			

Table 9 – Pairwise comparisons of environmental sub-criteria with respect to environmental criteria

	EN1	EN2	EN3	EN4
EN1	1.0	1	1	3.0
EN2		1.0	1.0	3.0
EN3			1.0	3.0
EN4				1.0

Table 10 – Pairwise comparisons of social sub-criteria with respect to social criteria

	S1	S2	S3	S4
S1	1.0	3.0	1.0	7.0
S2		1.0	½	2.0
S3			1.0	5.0
S4				1.0
Inconsistency: 0.01				

Table 11 – Relative values of remediation technologies and sustainability criteria from the literature and technical documentations (after CLU-IN c.2010, FRTR c 2010, Friend, Air force)

	Direct costs	Indirect costs	Impacts on other resources	Impacts on ecological systems	Intrusive.	Resource use and waste by-products	Impacts on human health	Impacts on neigh. land	Uncertainty, evidence and fit with policy
Bioventing	Low	Low	Average	Average	Average	Low	Low	Low	High
Enhanced bioremediation	Average	High	Low	Average	Low	Average	Low	Low	High
Monitored natural attenuation	Low	High	Low	Low	Low	Low	Low	Low	High
Phytoremediation	Low	Low	Low	Low	Low	Average	Low	Low	High
Air sparging	Low	Low	Average	Average	Average	Average	Low	Low	High
Soil vapour extraction	Average	High	Average	Average	Average	Average	Low	Low	High
Thermal treatment	High	High	High	Average	Average	High	Low	Low	High
Soil washing	High	High	High	High	High	High	Low	Low	High
Incineration	High	High	High	High	High	High	Low	Low	High
Thermal desorption	High	High	High	High	High	High	Low	Low	High
Excavation and disposal	Low	Low	High	High	High	Average	Low	Low	High

Table 11 – Pairwise comparisons of remediation technologies with respect to direct costs

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	3.0	1.0	1.0	1.0	3.0	5.0	5.0	5.0	5.0	1.0
A2		1.0	1/3	1/3	1/3	1.0	3.0	3.0	3.0	3.0	1/3
A3			1.0	1.0	1.0	3.0	5.0	5.0	5.0	5.0	1.0
A4				1.0	1.0	3.0	5.0	5.0	5.0	5.0	1.0
A5					1.0	3.0	5.0	5.0	5.0	5.0	1.0
A6						1.0	3.0	3.0	3.0	3.0	1/3
A7							1.0	1.0	1.0	1.0	1/5
A8								1.0	1.0	1.0	1/5
A9									1.0	1.0	1/5
A10										1.0	1/5
A11											1.0
Inconsistency 0.01											

Table 12 – Pairwise comparisons of remediation technologies with respect to indirect costs

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	5.0	5.0	1.0	1.0	5.0	5.0	5.0	5.0	5.0	5.0
A2		1.0	1.0	1/5	1/5	1.0	1.0	1.0	1.0	1.0	1.0
A3			1.0	1/5	1/5	1.0	1.0	1.0	1.0	1.0	1.0
A4				1.0	1.0	5.0	5.0	5.0	5.0	5.0	5.0
A5					1.0	5.0	5.0	5.0	5.0	5.0	5.0
A6						1.0	1.0	1.0	1.0	1.0	1.0
A7							1.0	1.0	1.0	1.0	1.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.00											

Table 13 – Pairwise comparisons of remediation technologies with respect to time span

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	1.0	1/3	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0
A2		1.0	1/3	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0
A3			1.0	1.0	3.0	3.0	3.0	5.0	5.0	5.0	3.0
A4				1.0	3.0	3.0	3.0	5.0	5.0	5.0	3.0
A5					1.0	1.0	1.0	3.0	3.0	3.0	3.0
A6						1.0	1.0	3.0	3.0	3.0	3.0
A7							1.0	3.0	3.0	3.0	3.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.01											

Table 14 – Pairwise comparisons of alternatives with respect to impacts on other resources

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1		1.0	1.0	1.0	3.0	3.0	5.0	5.0	5.0	5.0	1.0
A2			1.0	1.0	3.0	3.0	5.0	5.0	5.0	5.0	1.0
A3				1.0	3.0	3.0	5.0	5.0	5.0	5.0	1.0
A4					3.0	5.0	5.0	5.0	5.0	5.0	1.0
A5						1.0	3.0	3.0	3.0	3.0	1/3
A6							3.0	3.0	3.0	3.0	1/3
A7								1.0	1.0	1.0	1/5
A8									1.0	1.0	1/5
A9										1.0	1/5
A10											1/5
A11											
Inconsistency 0.01											

Table 15 – Pairwise comparisons of alternatives with respect to impacts on ecological systems

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	1.0	1/3	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0
A2		1.0	1/3	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0
A3			1.0	1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0
A4				1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0
A5					1.0	1.0	1.0	3.0	3.0	3.0	3.0
A6						1.0	1.0	3.0	3.0	3.0	3.0
A7							1.0	3.0	3.0	3.0	3.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.01											

Table 16 – Pairwise comparisons of alternatives with respect to impacts on intrusiveness

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1		1/3	1/3	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0
A2			1.0	1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0
A3				1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0
A4					3.0	3.0	3.0	5.0	5.0	5.0	5.0
A5						1.0	1.0	3.0	3.0	3.0	3.0
A6							1.0	3.0	3.0	3.0	3.0
A7								3.0	3.0	3.0	3.0
A8									1.0	1.0	1.0
A9										1.0	1.0
A10											1.0
A11											
Inconsistency 0.01											

Table 17 – Pairwise comparisons of alternatives with respect to resource use and waste by-products

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	3.0	1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0	3.0
A2		1.0	1/3	1.0	1.0	1.0	3.0	3.0	3.0	3.0	1.0
A3			1.0	3.0	3.0	3.0	5.0	5.0	5.0	5.0	3.0
A4				1.0	1.0	1.0	3.0	3.0	3.0	3.0	1.0
A5					1.0	1.0	3.0	3.0	3.0	3.0	1.0
A6						1.0	3.0	3.0	3.0	3.0	1.0
A7							1.0	1.0	1.0	1.0	1/3
A8								1.0	1.0	1.0	1/3
A9									1.0	1.0	1/3
A10										1.0	1/3
A11											1.0
Inconsistency 0.01											

Table 18 – Pairwise comparisons of alternatives with respect to impacts on human health

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A2		1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A3			1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A4				1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A5					1.0	1.0	1.0	1.0	1.0	1.0	1.0
A6						1.0	1.0	1.0	1.0	1.0	1.0
A7							1.0	1.0	1.0	1.0	1.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.00											

Table 19 – Pairwise comparisons of alternatives with respect to neighbouring land use

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A2		1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A3			1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A4				1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A5					1.0	1.0	1.0	1.0	1.0	1.0	1.0
A6						1.0	1.0	1.0	1.0	1.0	1.0
A7							1.0	1.0	1.0	1.0	1.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.00											

Table 20 – Pairwise comparisons of alternatives with respect to uncertainty, evidence and policy

	A1	A2	A3	A4	A5	A6	A7	A8	A9	A10	A11
A1	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A2		1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A3			1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
A4				1.0	1/3	1/3	1/3	1/3	1/3	1/3	1/3
A5					1.0	1.0	1.0	1.0	1.0	1.0	1.0
A6						1.0	1.0	1.0	1.0	1.0	1.0
A7							1.0	1.0	1.0	1.0	1.0
A8								1.0	1.0	1.0	1.0
A9									1.0	1.0	1.0
A10										1.0	1.0
A11											1.0
Inconsistency 0.00											

After all the comparison matrices have been completed and their consistencies checked with the EC software, the overall priority of the alternatives was derived by aggregating the local priorities across all criteria using (Ishizaka and Labib 2009):

$$p_i = \sum_j w_j * l_{ij}$$

Where p_i is the overall priority of alternative i,

l_{ij} the local priority, *and*

w_j weight of the criterion j.

Two approaches are used for deriving global priorities: (*i*) the ideal mode; *and* (*ii*) the distributive mode, which do not necessarily provide the same ranking. The ideal mode normalises by dividing the score of each alternative only by the score of the best alternative under each criterion. This prevents rank reversal and is suited for decision models that might change with addition and/or deletion of criteria or alternatives. Rank reversal occurs when judgements are altered when alternatives are added or deleted, even when the additions are irrelevant and deletion does not result in loss of information. Advocates of utility theory argue that adding alternatives, even irrelevant ones, should not cause rank reversal (Saaty 1990). AHP proponents however consider rank reversal as an asset as it mirrors normal human behaviour. Moreover, rank reversal phenomenon is not unique to AHP but to all additive models (Ishizaka and Labib 2009).

The distributive mode is suitable when the priorities are known. However, rank reversal may occur when alternatives are added and/or deleted, even if they are a copy of an existing alternative. Because the decision model might change with the addition and/or deletion of new criteria or alternatives, the ideal mode was used in the EC software so as to prevent rank reversal. The global priorities of the alternatives in the ideal and the distributive modes are presented in *Figures 2* and *3* respectively. As can be seen from the diagrams, although the priorities are different, the ranking is almost identical, with the exception of thermal treatment and excavation and disposal ranks reversing.

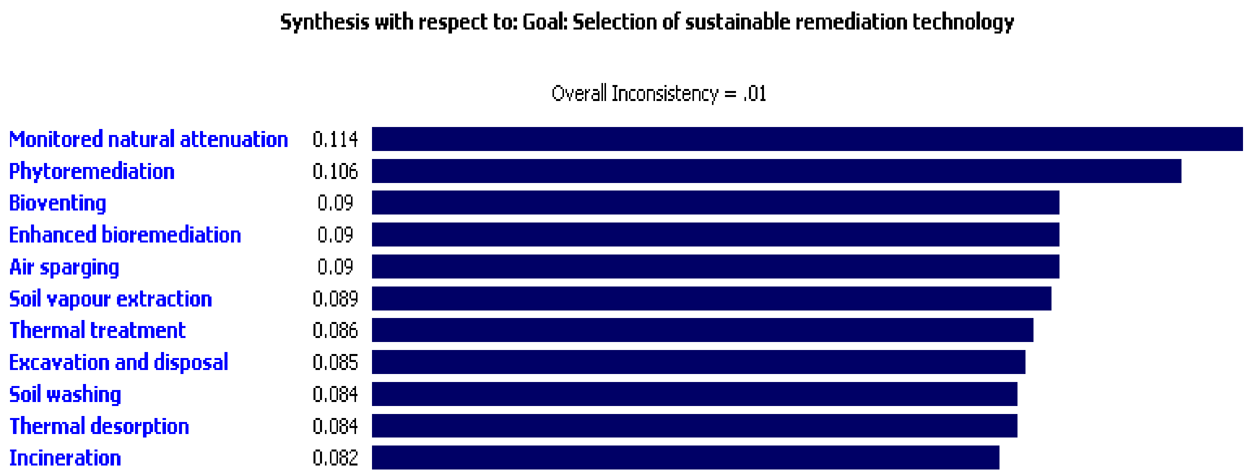

Figure 2 – Global priorities in ideal mode

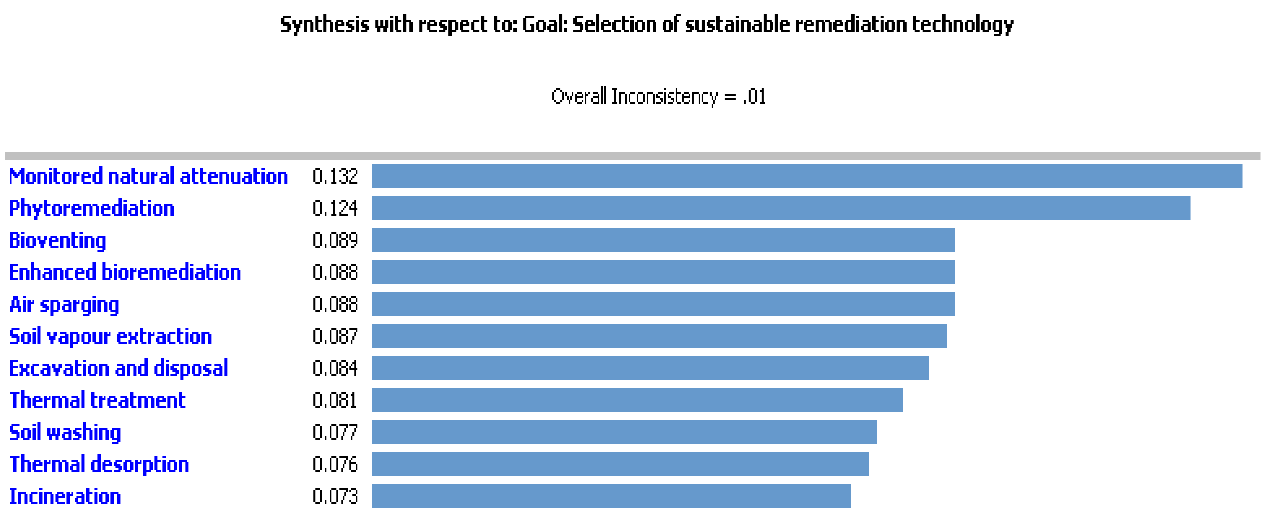

Figure 3 – Global priorities in distributive mode

A sensitivity analysis was carried out to check the effects of changing the criteria weights on the priorities (*Fig 4*). The results are said to be robust if the ranking does not change. Only a slight change in ranking is observed with increasing either the weights of the social or environmental criteria. A significant change in ranking most observed with increasing the weight of the economic criterion. No change in ranking was observed with reducing the weights of the social criterion and only a slight change was observed with reducing the weight of the economic criterion. Reducing the environmental criterion also resulted in significant change in ranking. Overall, the most robust criterion is the social criterion as changing its weight resulted in the least change in ranking of the alternatives.

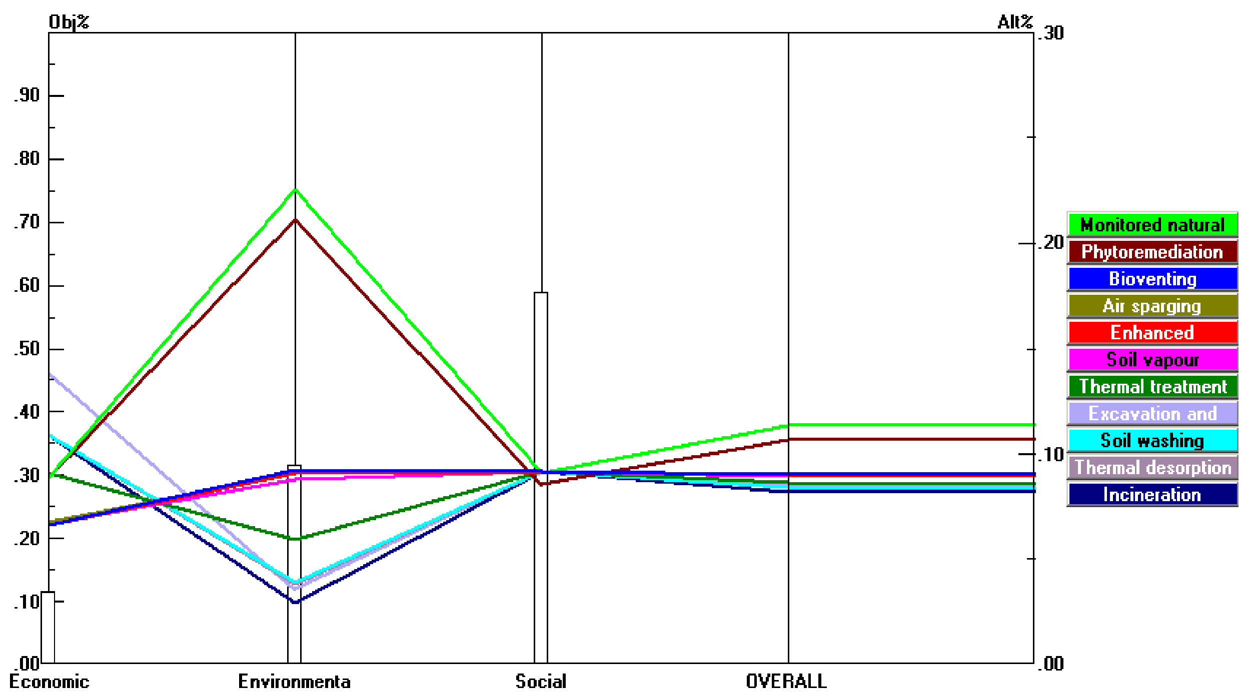

Figure 4 – Sensitivity graph displaying the performance of alternatives perform with respect to criteria.

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