2012 | Faculteit Bedrijfseconomische Wetenschappen

DOCTORAATSPROEFSCHRIFT

The cost effectiveness of gentle remediation strategies considering technical uncertainty and reversibility

Proefschrift voorgelegd tot het behalen van de graad van doctor in de toegepaste economische wetenschappen te verdedigen door:

Tine Compernolle

Promotor: prof. dr. ir. Steven Van Passel

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Bedankt. Heel erg bedankt.

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"Hoe..? Waarom, …? Maar, …?"
"Waar komt die 'B' ineens vandaan?"
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Summary

Environmental resources are used not only as an input to our production processes, but also as a sink for the wastes of these processes. Groundwater quality is threatened by urbanization, industrial development, and agricultural activities. Once groundwater is contaminated, it may aversively affect human health, degrade the environment, render land unfit for reuse, cause public concern, and erode quality of life.

To deal with groundwater contamination, a few basic steps should be followed. First, the site needs to be characterized properly and the risk associated with the contamination should be assessed. Based on these findings, a remediation objective is set. Then, different remediation approaches can be evaluated and compared. After a testing phase, the selected remediation strategy can be implemented on the field. By taking groundwater samples, it is possible to follow up the remediation progress and modify if necessary.

Focus of this dissertation is the application of phytoremediation and bioremediation as a possible approach for groundwater remediation. Phytoremediation is the use of plants and their associated bacteria to degrade, sequester, or volatilize contaminants. The bioremediation strategies considered in this dissertation are natural attenuation and biostimulation. Whereas natural attenuation relies on natural processes of degradation, biostimulation involves the intentional stimulation of the contaminant degradation achieved by the addition of water, nutrients, electron donors, or acceptors.

Within this dissertation, the cost effectiveness of phytoremediation and bioremediation is evaluated by using a multidisciplinary approach in which hydrogeological, technical, biological, and economic aspects are integrated. Besides this economic evaluation, it is also shown how firms responsible for clean-up can deal with variability within bio-physical processes, resulting in an uncertain contaminant removal efficiency and hence, an uncertain remediation time. By integrating the option to abandon phytoremediation or bioremediation when the application proves to be inefficient, the value of the remediation

increases. Integrating the option to abandon stimulates the adoption of these gentle groundwater remediation strategies.

Chapter 1 presents the techno-economic framework for groundwater remediation. It introduces the different steps that should be taken to realize site clean-up and the different remediation strategies considered in this dissertation. The following different problem statements are formulated: (i) how should the cost effectiveness of different remediation alternative be evaluated? (ii) Is the use of groundwater models valuable? (iii) How to integrate all the different aspects of groundwater remediation? (iv) Can technical uncertainty be integrated by the use of a decision tree? (v) Is it possible to apply the real options theory to define an optimal timing to abandon a remediation strategy?

The different methodologies applied are listed in **Chapter 2**. Concerning the cost effectiveness analysis, both the average and incremental cost effectiveness analysis are described. It is also shown how both a decision tree and the real options theory can be applied to integrate the option to abandon a remediation strategy within the economic analysis. An overview of different valuation methods for groundwater remediation is provided as well. Besides these economic methods, also different hydrogeological concepts which are relevant for this study are described.

Chapter 3 shows the results of the cost effectiveness analyses for both phytoremediation and bioremediation. It is demonstrated that when an incremental cost effectiveness ratio is calculated, the decision maker is more informed to select a remediation strategy. This ratio better defines the trade-off between costs and effects compared to the average cost effectiveness ratio. For the cases studied, both phytoremediation and bioremediation can be considered as cost effective alternatives among the available remediation strategies. Whether phytoremediation or bioremediation should be selected, depends on the benefit of the remediation. Moreover, it is shown that when the location at which the effectiveness is measured differs, or when the effectiveness is measured in another unit, the economic results will differ. Besides the economic evaluation of the different remediation strategies, also the value of hydrogeological modeling is determined. It is demonstrated that the correct use of groundwater flow and transport models can lead to better decision-making resulting in the application of more effective and less expensive groundwater remediation strategies.

The value of the option to abandon the remediation strategy, if it fails to remove the contamination sufficiently, is first evaluated by the development of a decision tree **(Chapter 4)**. Although this analysis shows that the introduction of this option increases the value of the remediation, it is not possible to determine the optimal timing at which the remediation strategy should be redirected. That is why the real option theory is applied **(Chapter 5)**. For the bioremediation case study, it is demonstrated that without the option to abandon, the bioremediation strategy would not be selected. For phytoremediation however, the remediation cost is much lower compared to conventional remediation strategies. Whether it takes 5 or 25 years for phytoremediation to reach the remediation objective, if only the sale value of the property is considered as the benefit of the remediation, phytoremediation is the preferred remediation strategy (compared to excavation).

Chapter 6 gives an overview of the main conclusions reached in this dissertation. Also a wider discussion on the main topics dealt with, is provided.

Samenvatting

Grondwater wordt niet enkel gebruikt als input voor productieprocessen, ook het afval van diezelfde processen komt terecht in grondwater. De kwaliteit van het grondwater wordt bedreigd door verstedelijking, industriële ontwikkeling en landbouwactiviteiten. Verontreinigd grondwater kan leiden tot negatieve gezondheidseffecten, een beperking van het landgebruik en publieke onrust.

Wanneer grondwater verontreinigd is, moeten verscheidene stappen gevolgd worden om het grondwater te zuiveren. Eerst moeten de karakteristieken van de site bepaald worden, evenals het risico dat de verontreiniging veroorzaakt. Gebaseerd op deze bevindingen wordt een saneringsdoelstelling opgelegd. Vervolgens worden verscheidene saneringsstrategieën met elkaar vergeleken en geëvalueerd. De geselecteerde saneringsstrategie kan eerst getest worden om nadien toegepast te worden op de verontreinigde site. Gedurende de sanering worden er stalen genomen om de saneringsvooruitgang te evalueren en het saneringsproces indien nodig te wijzigen.

Deze doctoraatsstudie richt zich op de toepassing van fytoremediatie en bioremediatie voor de sanering van grondwater. Fytoremediatie is het gebruik van planten en hun bacteriën om de verontreiniging af te afbreken, te fixeren of te vervluchtigen. Natuurlijke attenuatie en biostimulatie zijn de vormen van bioremediatie die beschouwd worden in deze studie. Bij natuurlijke attenuatie wordt er uitgegaan van de natuurlijke processen reeds aanwezig in de ondergrond om het grondwater te saneren. In het geval van biostimulatie wordt biodegradatie van de verontreiniging gestimuleerd door het toevoegen van water, nutriënten, elektron donors of acceptors.

De kosteneffectiviteit van fytoremediatie en bioremediatie wordt vanuit een multidisciplinaire benadering onderzocht. Hierbij worden hydrogeologische, technische, biologische en economische aspecten geïntegreerd. Bovendien wordt er aangetoond hoe bedrijven die verantwoordelijk zijn voor de sanering, kunnen omgaan met de variabiliteit eigen aan bio-fysische processen. Deze variabiliteit leidt tot een onzekere verwijderingsefficiëntie van de verontreiniging, waardoor men ook niet zeker is van de saneringsduur. Door een bedrijf de mogelijkheid te geven fytoremediatie of bioremediatie stop te zetten indien blijkt dat er

onvoldoende verontreiniging verwijderd wordt, stijgt de waarde van de sanering. De integratie van deze optie in het economisch model stimuleert de adoptie van deze niet-ingrijpende saneringsstrategieën.

Hoofdstuk 1 introduceert het technisch-economisch kader voor grondwatersanering. Hierin worden de verschillende stappen die genomen moeten worden om te komen tot een sanering, voorgesteld. Verder wordt er ook een overzicht gegeven van de verschillende onderzoeksvragen en probleemstellingen waarop deze studie een antwoord tracht te bieden. Fytoremediatie wordt namelijk vaak beschouwd als een kosteneffectieve saneringsstrategie, zonder dat dit aangetoond wordt met een correcte economische analyse. Een dergelijke analyse zou erop gericht moeten zijn om de verscheidene aspecten van een sanering te integreren, hierbij gebruik makend van grondwatermodellen. Daarnaast moet ook rekening gehouden worden met onzekerheden. De onzekerheid die beschouwd wordt in deze thesis is de moeilijkheid om de saneringsdoelstelling te bereiken.

De verschillende methoden die toegepast worden in deze studie, zijn voorgesteld in **Hoofdstuk 2**. Wat betreft de kosteneffectiviteitsanalyse, is de berekening van zowel de gemiddelde als de incrementele kosteneffectiviteitsratio beschreven. Er wordt ook aangetoond hoe een beslissingsboom en de reële optie analyse toegepast kunnen worden om een optie tot stopzetting van een saneringsstrategie te integreren in de economische analyse. Daarnaast wordt een overzicht van verscheidene methoden voor de waardering van grondwatersanering gegeven. Dit hoofdstuk sluit af met een omschrijving van de hydrogeologische concepten relevant voor deze studie.

Hoofdstuk 3 stelt de resultaten van de kosteneffectiviteitsanalyse van zowel fytoremediatie als bioremediatie voor. Er wordt aangetoond dat wanneer de incrementele kosteneffectiviteitsratio berekend wordt, de beslissingsnemer beter geïnformeerd is om een bepaalde strategie te selecteren. Deze ratio slaagt er beter in om de afweging tussen kosten en effectiviteit weer te geven, in vergelijking met de gemiddelde kosteneffectiviteitsratio. Wat betreft de cases die bestudeerd zijn, kunnen zowel fytoremediatie als bioremediatie beschouwd worden als kosteneffectieve saneringsalternatieven die een plaats kunnen innemen naast andere beschikbare saneringsstrategieën. Of fytoremediatie of bioremediatie geselecteerd kunnen worden, hangt af van de waarde van de sanering. Bovendien wordt er aangetoond dat wanneer de effectiviteit van een saneringsstrategie verschillend bepaald wordt, ook de te verkiezen strategie kan verschillen. Naast de economische evaluatie van de verschillende saneringsstrategieën wordt ook de waarde van een hydrogeologisch model bepaald. Het gebruik van een grondwatermodel leidt tot de toepassing van meer effectieve en goedkopere saneringen.

De onzekerheid van biodegradatie en de waarde van de optie tot stopzetting van de saneringsstrategie bij toepassing van bioremediatie worden eerst onderzocht aan de hand van een beslissingsboom **(Hoofdstuk 4)**. Hoewel deze analyse aangeeft dat een dergelijke optie de waarde van een sanering doet stijgen, bleek het niet mogelijk om een precies tijdstip tot omschakeling te bepalen. Daarom wordt er in **Hoofdstuk 5** overgegaan op de toepassing van de reële optietheorie voor zowel bioremediatie als fytoremediatie. Wat betreft de case waarbij bioremediatie toegepast is, blijkt dat deze optie nodig is. Zonder deze optie zou bioremediatie niet geselecteerd worden. In tegenstelling tot bioremediatie is fytoremediatie goedkoper. Dit maakt dat wanneer alleen de verkoopprijs van de grond beschouwd wordt, het niet relevant is dat de saneringsduur bij fytoremediatie 5 of 25 jaar kan duren, fytoremediatie is sowieso te verkiezen (in vergelijking met bodemontgraving).

Hoofdstuk 6 geeft een overzicht van de conclusies en voorziet in een bredere discussie.

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1 Introduction

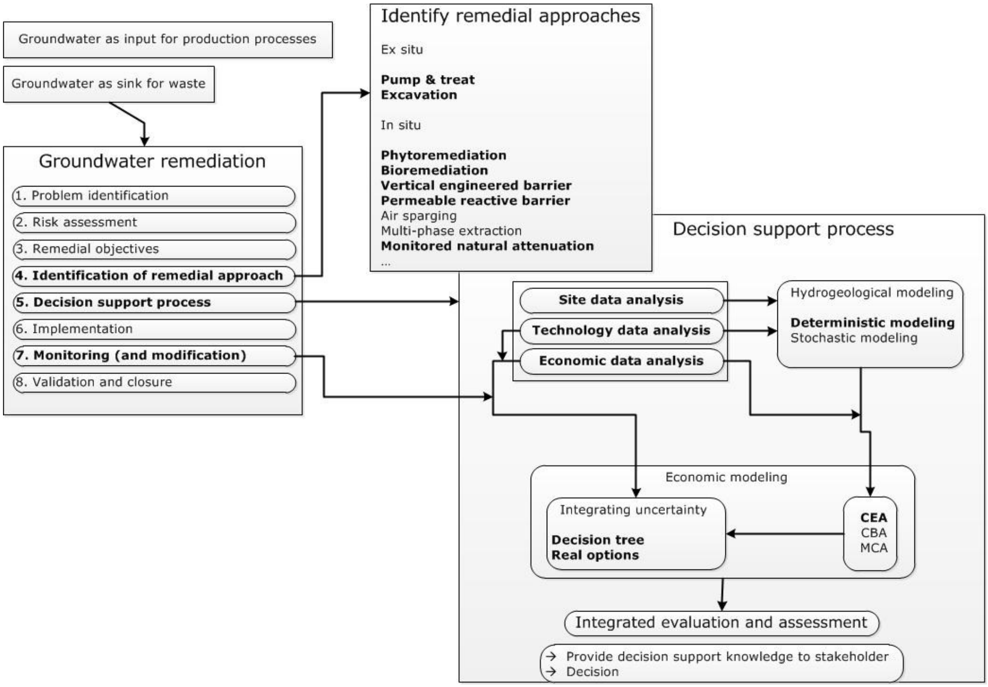


Figure 1.1 Framework for groundwater remediation. Based on Bardos *et al.* [4] and Hardisty, P.E. and Özdemiroglu, E. [3]

Environmental resources are used not only as an input to our production processes, but also as a sink for the wastes of these processes. Groundwater is such a natural resource that is not only used for domestic, agricultural or industrial purposes, it also contributes to surface water resources and plays a key role in the hydrologic cycle for the existence of ecosystems and natural beauty [1, 2]. However, urbanization, industrial development, and agricultural activities pose a threat to groundwater quality: fuel storage tank leakages, accidental spills and the excessive use of pesticides are only a few of all sources contaminating groundwater [2]. Groundwater contamination may aversively affect human health, degrade the environment, render land unfit for reuse, cause public concern and erode quality of life. If these effects are considered unacceptable, remedial action is required [3]. Figure 1.1 presents a framework that structures the different elements relevant for the remediation of groundwater contamination. The different aspects that this dissertation deals with, are highlighted.

1.1 Groundwater remediation

To approach a groundwater contamination problem, Hardisty and Özdemiroglu [3] define a few basic steps that should be followed. First, the site needs to be characterized properly in order to know the groundwater regime and the distribution and concentration of contaminants. Secondly, to understand the impact of adverse effects of groundwater contamination (such as human health, on the environment and public relations), the risk associated with the contamination should be assessed. Based on the site characteristics and the risk assessment, a remedial objective can be set. This objective can be the result of (i) a cost benefit analysis that justifies the maximum expenditure necessary to achieve the objective and (ii) a time constraint. Once the objective is set, various possible remedial approaches are evaluated and compared. The selected approach should be the one which is best able to reach the objective set within the applicable constraints. The selected remediation approach is then tested on a small scale before its implementation on the field. By taking groundwater samples, it is possible to follow up and modify the remedial progress if necessary. Once the objective is reached, it should be confirmed by the responsible authority after which the remediation project can be considered finished.

1.2 Groundwater remedial approaches

The treatment of contaminated soil and groundwater can be performed *ex situ* or *in situ*. Pump and treat (P&T) is the most commonly implemented groundwater remediation technology to treat groundwater [5]. This *ex situ* treatment involves the extraction of groundwater and an aboveground decontamination prior to discharge or reinjection. Physical, chemical, or biological treatment processes are applied to reduce contaminant concentrations to an acceptable level [6-10]. Bioremediation, air sparging, chemical treatment, and permeable reactive barriers (PRB) are the most commonly applied *in situ* treatment technologies [5].

This dissertation focuses on the application of phytoremediation and bioremediation for groundwater remediation.

Phytoremediation is the use of plants to degrade, sequester or volatilize contaminants. To successfully apply phytoremediation, root proximity, plant tolerance to contamination, and sufficient growth rates are conditions to be primarily fulfilled [11-18]. Despite these conditions, compared to traditional remediation technologies, phytoremediation is considered as a more environment-friendly technology that preserves soil fertility and structure. Due to high evapotranspiration rates, rapid growth, and phreatophytic root development, poplar trees are considered as ideal plants to remediate groundwater [18-20]. Poplars are inexpensive solar powered pumps that serve as hydraulic barriers [11, 15, 21, 22]. Even though phytoremediation is reported as being a cost effective remediation technology, as of 2011 only a limited number of phytoremediation projects were implemented. The USEPA (2010) reports 17 U.S. sites at which phytoremediation is applied for groundwater remediation, which is 5% of all in situ groundwater treatment projects. Also in the European Union, the use of phytotechnologies is limited [23].

Megharaj *et al.* [24] classify *in situ* bioremediation as bioattenuation, biostimulation and bioaugmentation. Whereas bioattenuation relies on natural processes of degradation, biostimulation involves the intentional stimulation of the degradation of chemicals achieved by addition of water, nutrients, electron donors, or acceptors. Bioaugmentation involves the addition of microbial members with proven capabilities of degrading or transforming the chemical pollutants. Like phytoremediation, bioremediation is considered as a cost effective remediation technology. However, factors such as low temperature, anaerobic conditions, low levels of nutrients and co-substrates, the presence of toxic substances, and the physiological potential of microorganisms can limit the efficiency of microbial degradation. Tyagi *et al.* [25] note that these drawbacks have resulted in a gap between laboratory trials and on-field application.

Problem statement 1

It is often highlighted that phytoremediation is an economical and effective remediation option compared to other remediation strategies [26-30]. However, often a thorough economic analysis for this statement is missing. In these reports, cost effectiveness is only determined on an average basis. It is not known whether total costs are discounted, the cost and effectiveness information provided is limited and no further conclusions are drawn. Or, comparisons do not seem to relate costs to the common site [26, 27] or only cost difference and no difference in remedial effect [28, 29] are reported. Linacre *et al.* [31] developed an economic decision model including the impact of uncertainty and came to some useful conclusions concerning the economic viability of phytoremediation. They demonstrated that property prices and technology development are important considerations for deciding whether phytoremediation or a more conventional remediation strategy should be applied. However, no real values for the remedial cost were used by Linacre *et al.* [31], which is critical for making meaningful cost comparisons.

Studies that define phytoremediation as being cost effective lack the use of an appropriate methodology. Also a correct comparison between phytoremediation and more conventional remediation strategies is missing. To make reliable conclusions on the cost effectiveness of phytoremediation, a correct methodology should be described and applied to specific case studies. This methodology should also be used to determine the cost effectiveness of a bioremediation strategy.

1.3 Identification of the remedial approach

The identification process of the preferred remediation technology applied in this dissertation is closely related to the framework characterized by Bardos *et al.* [4]. They describe a decision support process that assists in the identification of the optimal remedial approach. Site specific data, site specific knowledge and information on the technological properties of different proposed remediation approaches are combined and analyzed to determine the effectiveness of these different remedial options. The use of computer software facilitates the data analysis and allows the decision maker to have a clear insight into the problem which is often complex. However, one should be aware that the output is only as good as the data and modeling assumptions used. The output of the computer model can be integrated into an economic analysis to evaluate the different remediation options. The analyst should ensure that the analysis is accurate and the output is in a form useful for decision making. The knowledge is supplied to the decision makers who then have to decide whether the information is sufficient to make the decision.

To successfully take remedial actions, it is important to understand the behavior of contaminants in groundwater. Groundwater travels through heterogeneous and complex geologic media which makes it difficult to predict the contaminant behavior within the groundwater. Contaminants within the subsurface can accumulate and remain there for a long period of time because these are bound to the soil or rock. When any of these contaminants are soluble in water, a dissolved phase can be formed. This phase can migrate away with groundwater flow and is subject to hydrodynamic dispersion (causing spreading) and adsorption (causing retardation). As contaminants move in groundwater, they are also subject to various biological processes that will effectively reduce concentrations over time. Through a combination of data, geostatistical interpolation and flow and transport models, the decision maker can understand the groundwater flow regime, delineate the contamination and predict the patterns of contaminant movement in groundwater [3].

Problem statement 2

The simulation of groundwater flow can be based on a discrete set of initial conditions and parameter values *i.e.* deterministic simulations or hydrogeological stochastic models can be applied. Stochastic models are considered to be more useful as these allow for decision making in an uncertain framework. Uncertainties associated with subsurface heterogeneity are quantified and the dynamics of flow and transport are addressed [32, 33]. However, in practice, stochastic approaches are limited [33-35]. The expense of data gathering [36] and the large computational effort required [37] are important reasons why stochastic modeling is not widely applied. Moreover, in certain cases the results of a deterministic model do not differ much from the result of a stochastic model [38-41].

In some countries, not only the use of stochastic models is limited, also deterministic models are rarely applied to design remediation strategies. For instance in Flanders (Belgium), contractors mostly design by experience, and consultants rely on rough-and-ready rules to determine well location and use an analytic approach to determine the number of wells and extraction rate [42].

To demonstrate how valuable the use of a groundwater model can be, it is not only necessary to look into the increased effectiveness of the designed remediation strategies, an economic valuation of the groundwater model should be made as well. By describing a framework that presents the trade-off between the costs of developing a groundwater model and its benefits, the merits of a groundwater flow and transport model can be valued.

1.4 Integrated evaluation and assessment

Different economic analyses can be applied to make an economic evaluation of the different remediation options available. Risk-based CBA in combination with multi-criteria analyses are increasingly being applied to decision making for remedial option selection [4, 43]. Lemming *et al.* [44] present a framework for an integrated economic decision analysis which combines remediation costs, health risk costs and potential environmental costs. The framework proposed by Khadam and Kaluarachchi [45] integrates probabilistic risk assessment and multi-criteria decision analysis into a comprehensive framework for subsurface contamination management. Sorvari and Seppälä [46], too, adopted a multi-criteria approach and structured the decision problem in a value tree. Besides risk, also the preference of different stakeholders can be incorporated in a multi-criteria analysis [47].

Problem statement 3

These different studies show the necessity to integrate different disciplines to make a thoroughly conclusion on the optimal remediation strategy to apply. To evaluate phytoremediation and bioremediation as a strategy for groundwater remediation, biological processes, hydrogeological parameters, environmental aspects, health risk assessment and economics should be integrated in one evaluation.

1.5 Monitoring and modification

After the start of the remediation process, the remedial progress should be assessed and modified if necessary. The degree of knowledge of the site characteristics determines the success of clean-up. Failure of cleaning up a contaminated site can be due to an inappropriate assessment of the uncertainty related to both pollutant properties (*e.g.* contaminant strength and location) and hydrogeological parameters (*e.g.* horizontal conductivity) [48, 49]. To evaluate phytoremediation or bioremediation as cost effective remediation strategies, also the uncertain relationships among biomass, contaminants and nutrients should be incorporated in the economic decision analysis [24, 25, 50].

One way to deal with uncertainty is to adopt a sensitivity analysis. This analysis shows how the variation of the value of key parameters affects the economic result. A Monte Carlo sensitivity analysis involves the specification of probability distributions for uncertain values of model input parameters. Multiple trials are the executed, taking each time a random draw from the distribution for each parameter. With each trial, the output is calculated for each set of specified values. After all trials have been executed, a probability distribution of the model output is obtained. This kind of analysis concludes that when the economic decision does not change when parameter values are varied, the result is robust. If the economic decision differs for different parameter values, the decision maker then has to express a judgment about which value is most likely. However, a sensitivity analysis by itself resolves nothing, it only shows the sensitivity of the economic result [51].

Problem statement 4

An economic model should be developed to address the removal efficiency uncertainty inherent to bio– and phytoremediation strategies. It should be studied whether the uncertainty about the biodegradation rate can be characterized in terms of a number of distinct contingencies. If it is possible to assign probabilities of occurrence to each of these contingencies, then uncertainty about the biodegradation rate becomes a problem of dealing with risk. Risk can be readily incorporated into the decision analysis through expected value analysis. A decision tree can then visualize the logical structure of the decision problem in terms of sequences of decisions and realizations of contingencies. The next question is then whether a decision tree achieves to determine the value of the option to redirect the remediation strategy if bioremediation or phytoremediation fails to remove the contamination sufficiently.

Bage *et al.* [52, 53] developed a model that considers the possibility of reducing uncertainty by both acquiring more information on the level of contamination and offering the decision maker the opportunity to reevaluate his decision and switch to a more appropriate technology. However, only two stages are considered, an optimal timing to redirect the remediation strategy is not defined.

The option approach described by Dixit and Pindyck [54] however, takes into account the possibility to redirect a decision made, and an irreversible investment is considered as a sunk cost. These authors illustrate that, under price uncertainty, the opportunity cost of investing immediately, rather than waiting and keeping open the possibility of not investing, is a significant component of the firm's investment decision. The option value increases with the sunk cost of the investment and with the degree of uncertainty over the future price. Dynamic programming is a general mathematical technology for the optimization of sequential decisions under uncertainty. A whole sequence of decisions is split into two components: the immediate decision and a valuation function that encapsulates the consequences of all subsequent decisions.

Problem statement 5

It is unexplored whether dynamic programming can be applied to define an optimal timing to abandon a remediation strategy which proves to be inefficient. Within this dissertation, the uncertainty considered is not related to market conditions but to the technical difficulty to reach the remediation objectives set. It should be determined when it is optimal to abandon the applied remediation strategy. Within this analysis, groundwater samples can be considered as a source of information that arrives at determined points in time. It should be demonstrated how this information can be integrated in the economic analysis and to which extent decision making is influenced by this information.

The different problem statements lead to the following central research question:

1.6 Central research question

How cost effective is the application of phytoremediation and bioremediation for soil and groundwater remediation compared to more conventional technologies taking into account technical uncertainty and the reversibility of the remediation strategy?

Questions to be addressed:

- Which methodologies can be applied to determine whether phytoremediation or bioremediation are cost effective remediation strategies? (Chapter 2)
- Are phytoremediation and bioremediation cost effective groundwater remediation strategies? (Chapter 3)
- How valuable is it to use hydrogeological groundwater modeling to assess the effectiveness of different remediation strategies? (Annex II chapter 3)

- How can technological uncertainty be integrated in the decision making process? (Chapter 2)
- How can a decision tree be applied to deal with technical uncertainty? (Chapter 4)
- What is the optimal timing to abandon the applied remediation strategy if it fails to remove the contamination sufficiently? (Chapter 5)

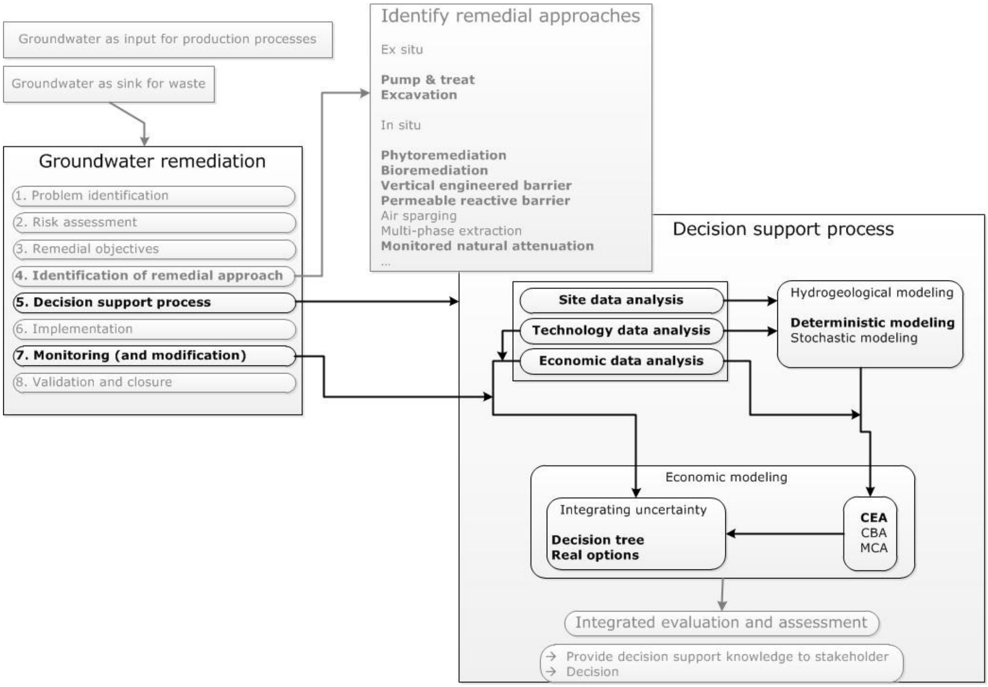
This dissertation deals with the identification of the remedial approach and the modifications necessary during the remediation process. More specifically, focus is put on the identification of phytoremediation or bioremediation as the preferred remediation technology among the broad range of remedial approaches available. It is demonstrated how technical variability can affect the selection process and how the possibility to modify the remediation approach can be incorporated into the decision making process.

The use of a cost effectiveness analysis to select a remediation technology is explored in depth. A distinction is made between an average cost effectiveness analysis that indicates the overall efficiency of a remediation strategy and an incremental cost effectiveness analysis that provides a true trade-off between cost and effects among the different remediation strategies available [55-57]. Using the latter economic evaluation tool, it is determined whether phytoremediation and bioremediation are cost effective groundwater remediation tools for specified contaminated sites.

The effectiveness of different groundwater remediation strategies is determined by a groundwater flow and transport model developed using the code MOCDENS3D in which MODFLOW is integrated to calculate groundwater flow [58, 59]. The output of this model is used in the economic evaluation of phytoremediation and bioremediation. Also the importance of using a hydrogeological model is discussed.

This dissertation further elaborates on the idea of redirecting the remediation as a means to address the removal efficiency uncertainty inherent to gentle remediation technologies, such as bioremediation and phytoremediation. Focus is put on the value of the option to redirect the remediation strategy. A decision tree is constructed to structure the different contingencies possible. An optimal timing to redirect the remediation strategy is determined by the application of dynamic programming.

2 Methodology



This chapter gives an overview of the methodologies used to evaluate the application of phytoremediation and bioremediation for different sites. A cost effectiveness analysis, a decision tree and the real option theory are applied to determine the economic impact resulting from the application of both technologies. A hydrogeological groundwater model is applied to determine how effective phytoremediation and bioremediation are for the remediation of contaminated groundwater compared to more conventional remediation strategies.

2.1 Cost effectiveness analysis

When a remediation objective is set for a specific contaminated site, there exist multiple strategies to achieve site clean-up. To determine which remediation strategy is preferable, a cost effectiveness analysis can be performed [51]. Two methods are distinguished: the average cost effectiveness ratio (ACER) and the incremental cost effectiveness ratio (ICER) [60]. The ACER determines the average cost per effect (C/E). Since benefits are not monetized, two different metrics are involved. While the cost of each technology is measured in monetary terms, the effect is measured in units, *e.g.* unit decrease in concentration. The computed ACERs can be ranked, forming a basis for deciding between the different technologies. The remediation strategy with the smallest ACER should be preferred. If the effectiveness of different remediation strategies is equal, the least costly strategy is chosen, or if all remediation technologies entail the same cost, the most effective clean-up is preferred [61, 62]. The ICER determines the ratio of the change in cost to the change in effect [55-57, 60-62] and is defined as $\Delta C/\Delta E$. ΔC represents the difference in cost between the alternative technology and the reference technology, ΔE represents the incremental effect.

A cost effectiveness plane like presented in Figure 2.1 is established to illustrate how decisions can be related to costs and effects [55, 56]. The four quadrants represent the relationship between the incremental cost and effect of each alternative remediation technology compared to the reference remediation technology. If case alternatives are located in quadrants II or IV, no trade-off between cost and effect takes place and the decision maker will decide to adopt the alternative or reference technology, respectively.

In case alternative remediation technologies are situated in quadrants I or III, making a decision is less straight forward. An increase in cost has to be balanced by an increase in effect (quadrant I) or a decrease in effect has to be balanced by a decrease in costs (quadrant III) [55-57, 62]. In order to find this balance, a monetary value (λ) should be attached to the incremental effect. This value represents the maximum price society is willing to pay for one more unit of remedial effect. If the monetary value of the change in effect is larger than the change in costs when an alternative remediation technology is applied, then the alternative technology should be preferred. When λ is known, the economic analysis is called a cost-benefit analysis instead of a cost effectiveness analysis. The net benefit (NB) equals $\lambda\Delta E - \Delta C$. If the NB is positive, the alternative technology should replace the reference technology. Figure 2.1 presents the relationship between the ICER and the net benefit [57].

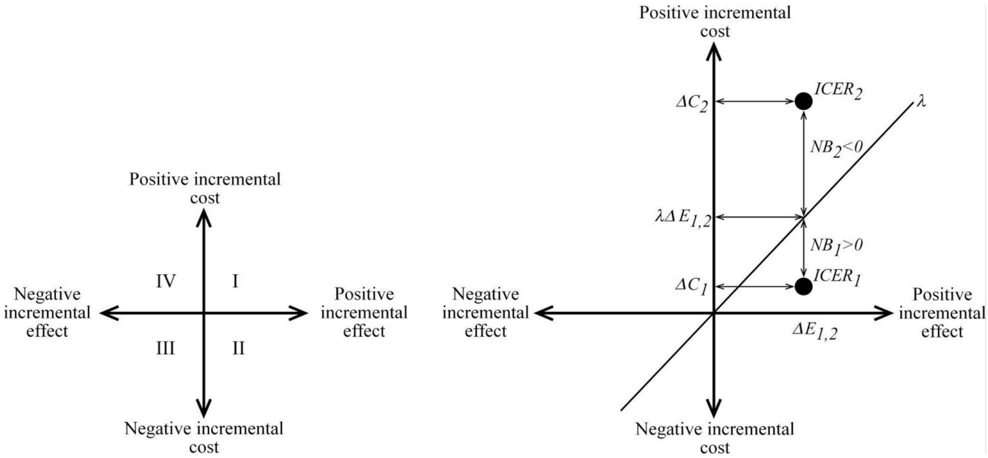


Figure 2.1 The cost effectiveness plane and the relationship between the incremental cost effectiveness ratio (ICER) and the net benefit (NB). The ICER of alternative 1 is situated under the willingness to pay line (λ). The monetary value of the incremental effect (λΔE) is larger than the incremental costs (ΔC_1). The net benefit (NB_1) is positive and the alternative remediation technology is preferred. The incremental cost of alternative 2 is larger than the monetary value of the incremental effect. NB_2 is negative and hence, the reference remediation technology should be applied.

2.2 Net present value and the design of a decision tree

The net present value (NPV) is an economic evaluation tool that supports decision making and that can be applied to select the appropriate remediation technology. In order to make a selection among the different remediation strategies available, the decision maker should adopt the strategy with the highest NPV. In order to determine the NPV of a remediation strategy the sum of the discounted annual net cash flows minus the investment cost are calculated as shown in Equation 2.1. The project year, the lifetime of the project, the discount rate, the annual net cash flows and the investment cost are represented by t, T, r, CF, and I respectively. The different factors that influence the annual cash flows are summarized by α.

$$NPV(r,a) = \sum_{t=1}^{T} \frac{CF_t(\alpha)}{[1+r]^{t-1}} - I \qquad (2.1)$$

The annual net cash flow involves the annual benefit minus the annual cost of the remediation strategies considered. To evaluate groundwater remediation technologies, the quantity of contaminant mass removed is considered as the annual benefit of a remediation strategy. Section 2.4 describes how the value of groundwater remediation can be determined.

The removal efficiency of remediation strategies such as bioremediation or phytoremediation is often uncertain. In order to deal with this uncertainty economically, an expected value analysis can be performed. Within this analysis, specific contingencies or 'states of the world' for the remediation strategy are determined. These contingencies represent the different possible levels of removal efficiency and can therefore be considered as exhaustive and mutually exclusive. The full range of likely variation is captured. Next, probabilities of occurrence are assigned to each of these contingencies. These probabilities should be nonnegative and sum to 1 [63]. To calculate the expected NPV of the remediation process, the NPV of the remediation process under each contingency is calculated and a weighted average of these NPVs over all the contingencies is taken. The weights are the respective probabilities that the contingencies will occur. For *n* contingencies, let NPV_i be the NPV under contingency *i* and p_i be the probability of contingency *i* occurring. In that case the expected NPV, *E[NPV]*, is given by Equation 2.2:

$$E[NPV] = p_1 NPV_1 + \cdots + p_n NPV_n \tag{2.2}$$

The calculation of the expected NPV does not take into account the reversibility of the remediation strategy. When the decision maker can decide to shift the remediation strategy at a certain point in time, the decision analysis is sequential: it proceeds in multiple stages. The logical structure of the decision problem in terms of sequences of decisions and realizations of contingencies can be represented by a diagram, which is called a decision tree. Within this diagram the initial decision is linked to final outcomes. Using backward induction, one works from final outcomes back to the initial decision, calculating expected NPVs across contingencies.

Figure 2.2 presents the decision tree which is used in this analysis to evaluate remediation processes which have an uncertain outcome. The decision maker has to decide whether to adopt a strategy of which the effectiveness is certain (strategy 1) or to adopt a remediation strategy of which the effectiveness varies (strategy 2). When the decision maker decides to adopt strategy 2, six possible contingencies exist: the contamination can be degraded (Degr.) at a rate of 0%, 20%, 40%, 60%, 80% or 100%. These contingencies occur with a probability of p_1, p_2, p_3, p_4, p_5 and p_6 respectively. At a certain point in time, the decision maker has the option to redirect the remediation strategy and to adopt strategy 1. Under each contingency, the NPV of continuing the remediation strategy is compared with the NPV of adopting strategy 1. The alternative with the highest NPV dominates and is used to calculate the expected NPV of strategy 2. Afterwards, the expected NPV of strategy 2 is compared with the NPV of strategy 1.

In order to determine whether it is valuable to have the option to redirect the remediation strategy, the expected NPV of strategy 2 with this option is compared with the expected NPV of strategy 2 without this option. When the latter expected NPV is calculated, all the NPVs of strategy 2 under the six contingencies are considered [NPV(1, STRAT2); NPV(2, STRAT2); … ; NPV(6, STRAT2)]. To determine the value of strategy 2 with option, for each contingency the outcome with the highest NPV is selected.

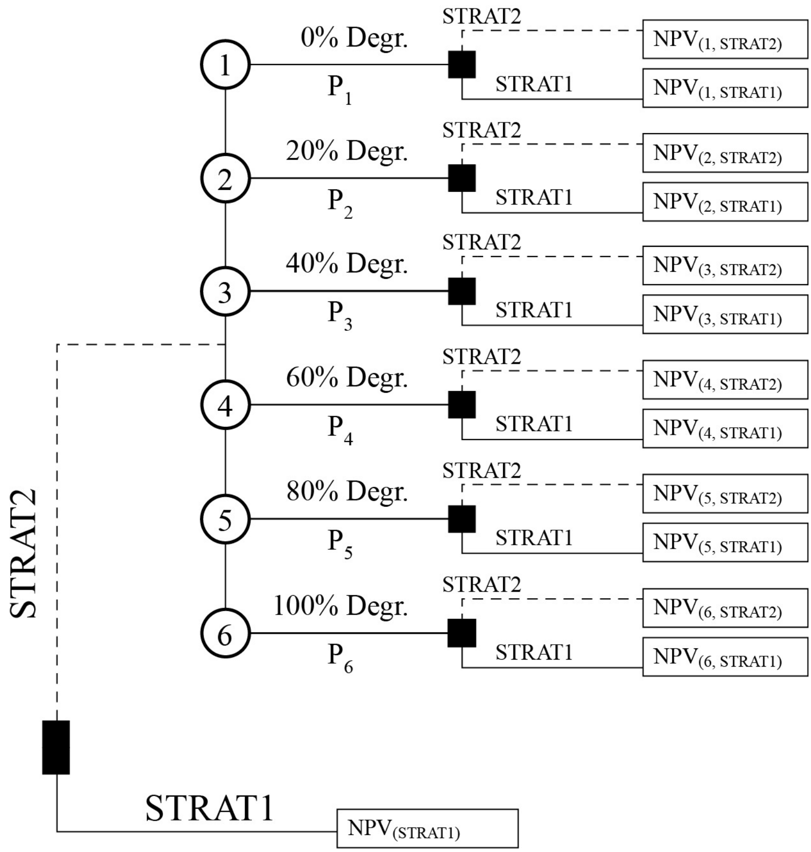


Figure 2.2 The decision tree for remediation strategy 1 with a certain effectiveness and remediation strategy 2 with an uncertain effectiveness

2.3 Real options analysis

The option approach is described by Dixit and Pindyck [54]. They developed the basic theory of irreversible investment under uncertainty, emphasizing the option-like characteristics of investment opportunities. Unlike the NPV theory, the option approach takes into account the possibility to redirect a decision made and an irreversible investment is considered as a sunk cost. These authors illustrate that under price uncertainty, the opportunity cost of investing immediately is a significant component of the firm's investment decision. The option value increases with the sunk cost of the investment and with the degree of uncertainty over the future price. Dynamic programming is a general mathematical technology for the optimization of sequential decisions under uncertainty. A whole sequence of decisions is split into two components: the immediate decision and a valuation function that encapsulates the consequences of all subsequent decisions.

For the economic evaluation of groundwater remediation strategies, a technical uncertainty is considered. This uncertainty relates to the physical difficulty to achieve the remediation standard: it is not known how much time, effort and materials will ultimately be required to meet the objectives set. This kind of uncertainty can only be resolved by undertaking the project [64]. In this study, a firm has invested in a remediation strategy of which the contaminant removal efficiency is uncertain (referred to as strategy 1). The firm faces the decision to continue the operation or to redirect the remediation strategy and adopt a remediation strategy of which the contaminant removal efficiency is certain (referred to as strategy 2). It is not known how effective strategy 1 will be. However, after strategy 1 starts operating, the efficiency is evaluated by taking samples, indicating the remediation strategy either to perform good or bad.

This type of problem closely relates to the studies performed by Jensen [65] and Thijssen *et al.* [66]. Jensen [65] describes a decision problem in which an innovation is introduced but the firm does not know whether the adoption will be profitable or not. By waiting and gathering information, this uncertainty can be resolved. This decision problem is formalized as an optimal stopping problem in which the firm can either stop waiting, adopt the innovative project and receive the expected return, or wait, learn from the observations and receive the expected value of this information. The firm starts with an initial belief that the innovation is profitable, which it updates each time it receives new information. The firm's learning behavior is assumed to be Bayesian: the belief is a conditional probability based on past information. While Jensen [65] only showed existence of a critical value of the belief in a good project at which investing is optimal, Thijssen *et al.* [66] extended this study and developed a framework in which an explicit expression is provided for this critical value.

Our study uses the framework of Thijssen *et al.* [66] to find the critical value of the belief in an inefficient groundwater remediation strategy at which the firm decides to stop its operation and adopt the strategy of which the removal efficiency is certain. Unlike the study of Thijssen *et al.* [66], the firm does not receive information if it waits and therefore has to decide whether or not to adopt strategy 1 immediately. This means that an initial value associated with the innovative project is included in the economic framework. If the firm decides to invest, it gathers information on the effectiveness (the revenues) of the remediation strategy by evaluating its performance. Based on that information, the firm decides to continue or adopt remediation strategy 2. Thijssen *et al.* [66] model the arrival of information, *i.e.* market signals via a Poisson process. Also this is different from our study. For our decision problem, samples which are taken at fixed moments in time are the source of information. Hence, only the quality of the signals, *i.e.* the probability with which the sample reflects the true state of the world is modeled as a binomially distributed random variable.

Consider a firm that has to meet certain remedial objectives and that can decide whether to invest in remediation strategy 1 under technical uncertainty or to invest in a strategy 2 with a certain removal efficiency. If the firm invests in strategy 1 and it proves to be inefficient, the firm can decide to abandon it and adopt strategy 2. The contaminant removal efficiency of the remediation strategy determines whether the application of the remediation strategy results in either a high revenue (U^H), or a low revenue (U^L).

Initially, the firm has a prior belief about strategy 1 being efficient or inefficient. The *ex ante* probability of the remediation strategy being efficient is given by

$$P(H) = p_0. \tag{2.3}$$

In Table 2.1, the first row lists the probabilities in case of an efficient remediation strategy (H) and the second row in case of an inefficient strategy (L). Hence, H and L represent the true state of the world in case of an efficient and inefficient strategy respectively. At determined moments in time, the firm evaluates the technological efficiency by taking a sample indicating the remediation strategy to be efficient (h-sample) or inefficient (l-sample). It is assumed that a correct sample always occurs with probability λ > ½.

Table 2.1 Probability of a sample indicating the remediation strategy to be efficient or inefficient, given the true state of the innovative strategy

	h	l
H	λ	1-λ
L	1-λ	λ

The samples are taken at fixed moments in time. The type of the sample is modeled as a binomially distributed random variable, indicating the remediation strategy either to be good or bad. Hence, denoting the number of h-signals by g, the dynamics of g is given by udn(t) = u, given dn(t) = 1,

$$u = \begin{cases} 1 \; with \; probability \; \lambda \text{ if H and } 1-\lambda \; if \; L, \\ 0 \; with \; probability \; 1-\lambda \; if \; H \; and \; \lambda \; if \; L, \end{cases}$$

and g(0) = 0. It is assumed that the firm knows the value of λ. The belief that revenues are high, *i.e.* that the remediation strategy is efficient, given the number of samples n and the number of h-samples, g ≤ n, is denoted by p(n,g). The conditional expected payoff of the uncertain remediation strategy can be written as

$$\mathbb{E}(U|n,g) = p(n,g)U^{H}_{Strategy\,1} + \big(1-p(n,g)\big)U^{L}_{Strategy\,1} - C_{Strategy\,1}, \tag{2.4}$$

In which $U^{H}_{\text{Strategy 1}}$ and $U^{L}_{\text{Strategy 1}}$ represent the incoming cash flow associated with a high and low technological efficiency. $C_{\text{Strategy 1}}$ is the outgoing cash flow which comprises the operational costs. To find the critical level at which the firm is indifferent between continuing remediation strategy 1 or adopt remediation strategy 2, while taking into account the option value of abandonment, first p(n,g) should be calculated. When this critical level is determined, it is known that it is optimal to adopt the strategy with an uncertain outcome as soon as p(n,g) is below this level. Define k = 2g – n, the number of good samples in excess of bad samples, and $\zeta = (1 - p_0)/p_0$, the unconditional odds of the remediation strategy being inefficient. By using Bayes' rule we obtain that [66]

$$p(n,g)=\frac{P(n,g|H)P(H)}{P(n,g|H)P(H)+P(n,g|L)P(L)}=\frac{\lambda^g(1-\lambda)^{n-g}p_0}{\lambda^g(1-\lambda)^{n-g}p_0+(1-\lambda)^g\lambda^{n-g}(1-p_0)}=\frac{\lambda^k}{\lambda^k+\zeta(1-\lambda)^k}\equiv p(k). \quad (2.5)$$

The critical level of k where the firm is indifferent between continuing the remediation strategy and redirecting the remediation process is denoted by $k^* \in \mathbb{R}$. At any arrival of an h-sample, k increases with unity and at any arrival of an l-sample, k decreases with unity. Hence, enough l-samples must arrive to reach the critical level. The critical level of the conditional belief in an efficient remediation strategy is denoted by $p^* = p(k^*)$ [66].

Suppose that the state of the process at a particular point in time is given by k. For the moment assume that k is a continuous variable. Then there are three possibilities. First, k might by such that $k \leq k^*$ and hence $p(k) \leq p^*$. Then it is optimal for the firm to directly abandon remediation strategy 1 and adopt remediation strategy 2. In this case, the value for the firm, denoted by $\boldsymbol{\Pi}_{\text{Strategy 2}}$, is given by

$$\Pi_{\text{Strategy 2}}=\sum_{t=0}^{\infty}\frac{\text{annual net cash flow}}{(1+r)^t}-I_{\text{Strategy 2}}\,. \quad (2.6)$$

A second possibility is that, even after an l-sample arrives it is not optimal to abandon the remediation strategy 1, *i.e.*, $k > k^*+1$. Because the time steps are discrete, the value function of the opportunity to adopt remediation strategy 2 for the firm, denoted by $V_1()$, must satisfy the following Bellman equation:

$$V_1(k)=\frac{1}{1+r}E(V_1(k')|k)+\Pi(k), \quad k>k^*+1. \quad (2.7)$$

We have that

$$\Pi\big(p(k)\big)=p(k)a+\big(1-p(k)\big)b, \quad (2.8)$$

$a = U^{H}_{Strategy1} - C_{Strategy\ 1}$; $b = U^{L}_{Strategy\ 1} - C_{Strategy\ 1}$,

and

$$E\big(V_1(k')\big|k\big) =$$
$$p(k)\big(\lambda V_1(k+1) + (1-\lambda)V_1(k-1)\big) + \big(1-p(k)\big)\big(\lambda V_1(k-1) + (1-\lambda)V_1(k+1)\big). \quad (2.9)$$

Substitution of Eq. (2.8) and Eq. (2.9) into (2.7) and rewriting gives

$$\begin{aligned}(1+r)V_1 = (1+r)\big(p(k)a + \big(1-p(k)\big)b\big) + \big(2p(k) + 1 - \lambda - p(k)\big)V_1(k+1) \\ + \big(p(k) + \lambda - 2\lambda p(k)\big)V_1(k-1). \quad (2.10)\end{aligned}$$

By substituting Eq. (2.5) the term associated with $V_1(k+1)$ can be written as

$$2p(k)\lambda + 1 - \lambda - p(k) = \frac{\lambda^{k+1} + \zeta(1-\lambda)^{k+1}}{\lambda^k + \zeta(1-\lambda)^k}. \quad (2.11)$$

Similarly, we can rewrite the term associated with $V_1(k-1)$ as

$$p(k) + \lambda - 2p(k)\lambda = \frac{\lambda(1-\lambda)(\lambda^{k-1} + \zeta(1-\lambda)^{k-1})}{\lambda^k + \zeta(1-\lambda)^k}. \quad (2.12)$$

Substituting Eq. (2.11) and Eq. (2.12) in Eq. (2.10) and defining $F(k) = \lambda^k + \zeta(1-\lambda)^k V_1(k)$, yields

$$(r+1)F(k) = (1+r)(a\lambda^k + b\zeta(1-\lambda)^k) + F(k+1) + \lambda(1-\lambda)F(k-1), \quad (2.13)$$

The solution of the homogenous equation is given by

$$F(k) = Am_1^k + Bm_2^k \quad , m_1 \neq m_2, \quad (2.14)$$

where A and B are constants and m_1 and m_2 are the roots of the homogenous equation

$$Q(m) \equiv -(r+1)m + m^2 + \lambda(1-\lambda) = 0. \quad (2.15)$$

Eq. 2.15 has two roots, namely

$$m_{1,2} = \frac{-b \pm \sqrt{D}}{2a} = \frac{(r+1)}{2} \pm \frac{\sqrt{(r+1)^2 - 4\lambda(1-\lambda)}}{2}. \tag{2.16}$$

Note that Q(m) is an upward pointing parabola (a>0) with $Q(0) = \lambda(1-\lambda) > 0$ and $Q(\lambda) = -r\lambda \leq 0$. Thus it holds that $m_1 > \lambda$ and $m_2 < \lambda$.

When the number of h-signals relative to l-signals tends to infinity, the value of the option to invest in strategy 2 converges to zero, *i.e.*

$$\lim_{k \to +\infty} \frac{\mathrm{A}m_1^k + \mathrm{B}m_2^k}{\lambda^k + \zeta(1-\lambda)^k} = 0. \tag{2.17}$$

This implies that we only need to consider the smaller root m_2, so that A=0.
The particular solution of Eq. (2.13) yields

$$\mathrm{F(k)} = \frac{\mathrm{a}\lambda^k(1+\mathrm{r})}{\mathrm{r}} + \frac{\mathrm{b}\zeta(1-\lambda)^k(1+\mathrm{r})}{\mathrm{r}}. \tag{2.18}$$

The value function V_1 is then given by

$$V_1(k) = \frac{rBm_2^k + (1+r)a\lambda^k + (1+r)b\zeta(1-\lambda)^k}{r(\lambda^k + \zeta(1-\lambda)^k)}. \tag{2.19}$$

A third and final possibility, is that $k^* < k \leq k^*+1$. The value of k is such that it is not optimal to invest in the certain remediation strategy right away. However, when the following sample is an l-sample, it will be optimal to abandon the uncertain remediation strategy. Analogous to Eq. (2.7) the Bellman equation for $V_2(k)$ is given by

$$V_2(k) = \frac{1}{1+r} E(V_2(k')|k) + \Pi(k), \;\; k > k^* + 1. \tag{2.20}$$

We have that

$$E\big(V_2(k')\big|k\big) = p(k)\big(\lambda V_1(k+1) + (1-\lambda)\Pi_{\text{strategy }2}\big) + \big(1-p(k)\big)\Big(\lambda\Pi_{\text{strategy }2} + (1-\lambda)V_1(k+1)\Big), \tag{2.21}$$

so that

$$V_2(k) = \frac{Bm_2^{k+1}}{(r+1)(\lambda^k + \zeta(1-\lambda)^k)} + \frac{(r+\lambda)a\lambda^k + (r+1-\lambda)b\zeta(1-\lambda)^k}{r(\lambda^k + \zeta(1-\lambda)^k)} + \frac{\lambda(1-\lambda)(\lambda^{k-1} + \zeta(1-\lambda)^{k-1})}{\lambda^k + \zeta(1-\lambda)^k}\frac{\Pi_{\text{Strategy }2}}{1+r}. \tag{2.22}$$

If an l-sample arrives, the process jumps to the region where k≤k* and if an h-sample arrives, the process jumps to the region where k>k*. Therefore, the value V_2 is completely determined by V_1(k+1) and $\Pi_{\text{strategy 2}}$. The value function V(k) then equals

$$V(k) = \begin{cases} V_1(k)\ if\ k >\ k^* + 1 \\ V_2(k)\ if\ k^* < k \leq k^* + 1 \\ \Pi_{strategy\ 2}\ if\ k \leq k^*, \end{cases}$$

where V_1(k), V_2(k) and $\Pi_{\text{strategy 2}}$ are given by equations (2.6), (2.19) and (2.22). To determine B and k* we solve the continuity condition $\lim_{\mathrm{k}\to\mathrm{k}^*+1} \mathrm{V}_1(\mathrm{k}) = \mathrm{V}_2(\mathrm{k}^*+1)$ and the value-matching condition $\lim_{\mathrm{k}\to\mathrm{k}^*} \mathrm{V}_2(\mathrm{k}) = \Pi_{\text{strategy }2}$.

The latter equation yields

$$B = \frac{\Pi_{\text{Strategy }2} r\big[\lambda^{k^*}(\lambda+r) - \zeta(1-\lambda)^{k^*}(\lambda-r-1)\big]}{\mathrm{r}m^{k^*+1}} - \frac{a\lambda^k\big(r+\lambda(1+r)\big) + b\zeta(1-\lambda)^k\big(r+(1-\lambda)(1+r)\big)}{\mathrm{r}m^{k+1}}. \tag{2.23}$$

Substituting B in the former equation leads to an expression for p* ≡ p(k*):

$$p(k^*) = \frac{X}{Y}, \tag{2.24}$$

where

$$X = \big(\mathrm{br} + \mathrm{b} - \Pi_{\text{Strategy }2} r\big)[(\lambda-1)^2 + m_2(\lambda-r-1) + r^2 - (\lambda-2)r],$$

$Y = m_2\big(a(r+1)(\lambda+r) - b(r+1)(-\lambda+r+1) + (1-2\lambda)\Pi_{\text{Strategy }2}r\big) +$
$(r+1)\big(-a(\lambda^2+r^2+\lambda r+r) + \mathrm{b}((\lambda-1)^2+\mathrm{r}^2-(\lambda-2)\mathrm{r}) + (2\lambda-1)\Pi_{\text{Strategy }2}r\big).$

The threshold number of h-signals relative to l-signals is then given by

$$k^* = \frac{ln\left(\frac{p^*}{1-p^*}\right) + \ln(\zeta)}{ln\left(\frac{\lambda}{1-\lambda}\right)}. \tag{2.25}$$

2.4 The benefits of groundwater remediation

Hardisty and Özdemiroglu [67] categorize the benefits of groundwater remediation into internal and external benefits. The internal or private benefits comprise the direct benefits like the sale value of a remediated site and avoided costs like avoided penalties. Public or external benefits are not included in the actual market transactions and correspond to the components of the total economic value (TEV) of groundwater. Hardisty and Özdemiroglu [1] define the TEV of groundwater as the sum of use and nonuse values. The use value includes the direct use value (*e.g.* value of domestic water use), indirect use value (*e.g.* value of the contribution to tourism) and, the option value (the value to have the option to use the groundwater). The nonuse value consists of the altruistic value (willingness to protect groundwater), existence value (value derived from the existence of clean groundwater), and bequest value (value derived from the desire to pass on the resource to future generations). Figure 2.3 presents the different benefit categories.

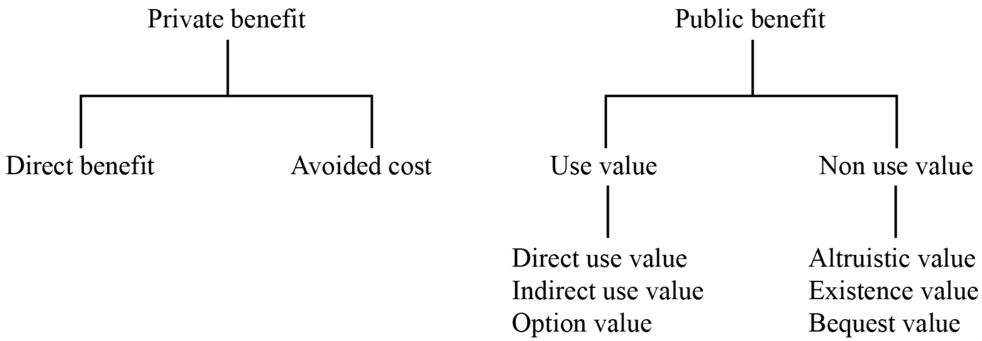


Figure 2.3 Overview of the groundwater benefit categories. Based on Hardisty and Özdemiroglu [1, 67]

Before the benefits of the remediation can be valued, the change in quality and quantity must be defined. A dose-response relationship links the change (dose) and the impacts of this change (response). Dose represents the type, magnitude, and route of contamination. Response is the impact of this contamination on the affected receptors (human health, the environment, various uses of groundwater).

To value the benefits of the remediation, different valuation techniques exist. Figure 2.4 presents an overview. Revealed preference techniques estimate that portion of the total economic value that is revealed in actual market transactions. People’s preference can be revealed through their behavior in actual and surrogate markets. The actual market price data used to estimate the value of groundwater includes potable water, irrigation water, and the value of land sold after clean-up [1]. Paleologos [68] calculates the lost value of groundwater due to pollution as the change of groundwater usage: from potable water, before a pollution event, to irrigation water, which is what is returned after. The calculation of avoidance costs, like the cost of illness infers benefits of good-quality environmental resources by measuring the consumption of goods and services that substitute for the environmental quality change [69]. Surrogate market techniques comprises the hedonic pricing technology (see Rosen [70] for a contribution to this literature) and the travel cost method. Guignet [71] investigates how people value environmental quality, by measuring impacts on home values from leaking underground storage tanks. It is revealed that respondents believe home prices decrease 18% to 24% when pollution is present. Also Young [69] describes how these methods can be applied to value water. Revealed preference techniques investigate markets that do not trade the environmental resource itself but are influenced by it. Stated preference techniques use questionnaires to determine how much individuals are willing to pay for environmental benefits or to avoid losses [1, 69]. For instance, Stengert and Willinger [72] estimate a mean willingness to pay for the preservation value of the groundwater quality for the Alsatian aquifer based on a contingent valuation study. Choice modeling is applied to determine the relative importance of groundwater characteristics and the willingness to pay for clean water. Lancaster [73] is the first author who does not consider goods as the direct object of utility, he states that it are the characteristics of the good from which utility is derived.

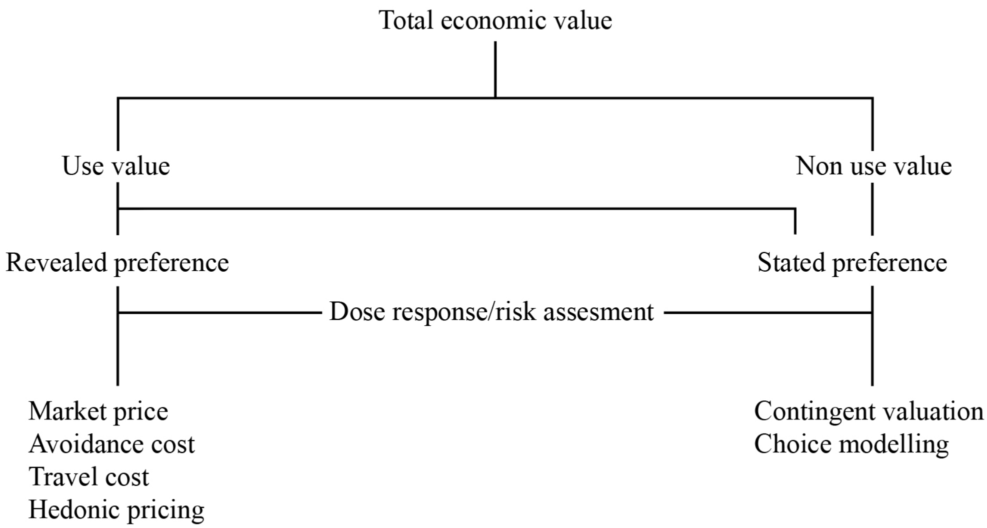


Figure 2.4 Overview of economic valuation techniques. Based on Hardisty and Özdemiroglu [1]

In this dissertation, the benefit of groundwater remediation is not a central topic. However in Chapter 3, the avoided health costs are estimated. To construct a decision tree, and for the real options analysis, a value for groundwater remediation needs to be determined. This value is based on the direct benefit, hence the sale value of the remediated site. For each case study, a sensitivity analysis is performed to evaluate the impact of variations of this value on the economic result.

2.5 Groundwater flow and transport modeling

This section briefly discusses the concepts of groundwater flow and the simulation of groundwater flow and solute transport. The information is primarily retrieved from the manual “Physical and chemical hydrogeology” written by P.A. Domenico and F.W. Schwartz [74].

2.5.1 Groundwater flow

The basis of groundwater flow is formed by the Darcy equation. Henry Darcy related the flow (Q) through a porous medium of a known cross-sectional area (A) to the applied hydraulic head difference (Δh) over a given distance and the hydraulic conductivity (K, a constant that is the function of the properties of the porous medium, characterizes the capacity of a medium to transmit water). K can vary over about 12 orders of magnitude with the lowest value for unfractured rocks and highest value for gravel. The hydraulic head is the water-level elevation measured with reference to a common datum.

$$\frac{Q}{A} = q = K * \left(\frac{\Delta h}{\Delta x}\right) \tag{2.26}$$

The specific discharge q is a volumetric flow rate per unit of surface area of the sample. Because water only moves to the pore openings making up the surface area, it is necessary to define a more realistic velocity v that is volumetric flow rate per unit area of connected pore space. The expression for v is

$$v = \frac{Q}{An_e} = \frac{q}{n_e} = \frac{K}{n_e} * \left(\frac{\Delta h}{\Delta x}\right) = \frac{Ki}{n_e} \tag{2.27}$$

With n_e representing the effective porosity. Total porosity is defined as the part of rock that is void space, expressed as a percentage. Effective porosity is defined as the percentage of interconnected pore space.

The hydraulic head provides the energy for groundwater flow. Groundwater flows from high to low total head. The dimensionless hydraulic gradient, i represents the change in water level elevation over a certain distance. As mentioned, individual geologic units commonly have very different hydraulic conductivities. To create an average hydraulic conductivity, the following equations should be applied:

$$K_x = \frac{\sum(m_i K_i)}{\sum m_i} \tag{2.28}$$

Where K_x is the equivalent horizontal conductivity, K_i is the homogenous conductivity of an individual layer and m_i is the thickness of the layer. K_z is an equivalent vertical conductivity for the layered system.

$$K_z = \frac{\sum m_i}{\sum(m_i/K_i)} \tag{2.29}$$

For horizontal flow, the most permeable units dominate the system, for vertical flow the least permeable units dominate the system. Horizontal flow is faster than vertical flow.

2.5.2 Groundwater flow simulation

To simulate groundwater flow, first a conceptual model should be created. This model encompasses the hydrogeologic framework (shape, thickness and hydraulic properties of the geologic unit, hydraulic head distribution and hydraulic recharge) and boundary conditions. Boundary conditions can be specific hydraulic head boundaries, like a river defined by constant hydraulic head values.

The response of an aquifer system to for instance the extraction of groundwater is calculated by the numerical solution to a ground water flow equation. In the groundwater flow equation, the unknown is hydraulic head. The main numerical approaches used in practice are finite-difference and finite-elements methods. In this dissertation, the finite-difference method is used. The finite-difference approach uses a regular discretization where an aquifer is subdivided into a series of rectangular grid blocks. In a three-dimensional model, individual units are subdivided vertically into cells of a specified thickness. Associated with the grid blocks are nodes that represent the points where the unknown hydraulic head is calculated. Every node or cell must be supplied with information on hydraulic conductivity, transmissivity, storativity, and fluxes due to recharge, pumping and evaporation. Also boundary conditions should be integrated. Figure 2.5 shows the discretization of a three dimensional system. The grid system is referenced in terms of a row, column, and layer-numbering scheme with block-centered nodes.

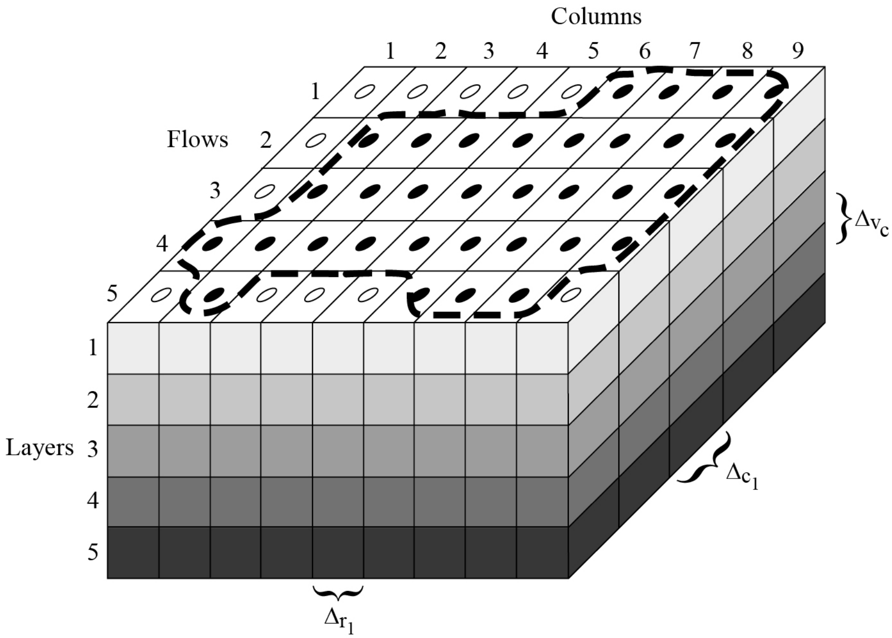


Figure 2.5 Discretization of a three dimensional system, indicating the rows, columns and layers of the model

2.5.3 Solute transport

Advection and dispersion are the two mass transport processes that spread dissolved mass in groundwater.

Advection is mass transport due simply to the flow of water in which the mass is dissolved. The factors that influence groundwater flow patterns (water table configuration, geologic layering, pumping) control the direction and rate of mass transport. When only advection is operating, mass added to one or more stream tubes will remain in those stream tubes. The direction of mass spreading in steady state systems is defined by path lines. Knowledge of groundwater flow patterns guides us in interpreting patterns of contaminant migration. Groundwater and the dissolved mass will move at the same rate and in the same direction. Accordingly, the velocity of advective transport is described by the Darcy equation (Eq. 2.23). However, when sorption occurs, the advective transport is less than that of groundwater.

Dispersion spreads some of the mass beyond the region it would occupy due to advection alone. There is spreading both ahead of the advective front in the same flow tube and laterally into the adjacent flow tubes. This dispersion is referred to as longitudinal and transverse (laterally and vertically) dispersion, respectively. Dispersion occurs in a porous medium because of diffusion and mixing due to velocity variations.

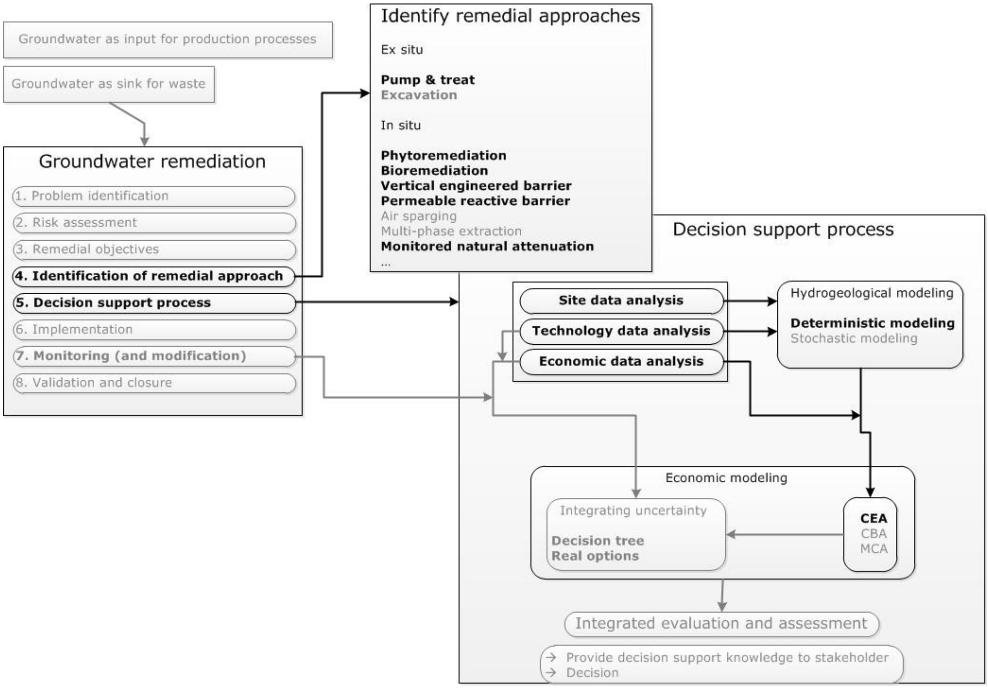
Another important process is a simple, first-order kinetic reaction that could account for biodegradation. The smaller the half-life of a kinetic reaction, the smaller the plume will be at a given time. Also this process is included in the hydrogeologic model.

2.5.4 The use of groundwater models

To know whether simulations are realistic, model results should be compared with measured values. Calibration is a process of selecting model parameters to achieve a good match between the predicted and measured hydraulic heads. Calibration is accomplished by a trial-and-error adjustment of model parameters. Models should be perceived as representations, useful for guiding further study but not susceptible to proof. The model design depends significantly on the informed judgment of its builder rather than on real information. Therefore, predictions made with simulation models must be interpreted with caution.

Within this dissertation, the groundwater flow and transport model was developed using the code MOCDENS3D in which MODFLOW is integrated to calculate groundwater flow [58, 59]. MODFLOW is the de facto standard code for aquifer simulation. A three-dimensional model is used to evaluate how effective phytoremediation and bioremediation are compared to other, more conventional groundwater remediation technologies. For each case study discussed, an overview of the hydrogeological properties applied in the groundwater model is given.

3 The cost effectiveness of phytoremediation and bioremediation[1]



This chapter demonstrates how the economic theory on cost effectiveness described in section 2.1 can be applied to select a groundwater remediation strategy. Two case studies are considered: one study regarding phytoremediation and the other regarding bioremediation. For each of these studies, the effectiveness of different remediation strategies is determined using groundwater modeling (see section 2.5). Cost calculations are based on

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[1] Parts of this section are published in: Compernolle, T., Van Passel, S., Weyens, N., Vangronsveld, J., Lebbe, L., Thewys, T. (2012). *Groundwater remediation and the cost effectiveness of phytoremediation.* International journal of phytoremediation, 14: 9, 861-877.

information provided by the companies responsible for groundwater remediation.

3.1 The cost effectiveness of phytoremediation

In this section the cost effectiveness of phytoremediation is compared with the cost effectiveness of other, more conventional remediation strategies. The case under consideration involves a site at which phytoremediation is actually applied. After the site description, the groundwater flow and transport model is presented. It is explained how the occurrence of the contamination and the application of the different remediation strategies are simulated. As regards the contaminant occurrence and phytoremediation, it is shown that simulated contaminant concentrations match observed concentrations. The cost and effectiveness of phytoremediation are compared to the cost and effectiveness of alternative remediation strategies by calculating the ACERs and ICERs as explained in Chapter 2. Further, it is demonstrated how a health impact assessment and the value of avoided health effects can be integrated in the analysis. Also the impact of a different hydraulic gradient on the cost and effectiveness of each remediation strategy is determined. At the end of this section, the conclusions drawn from these analyses are presented.

3.1.1 Contaminant site description

This site under consideration belongs to a Belgian car factory and is bordered by the Albert Canal to the north by northeast and the creek ‘Kaatsbeek’ to the south by southwest. The geological structure consists of a 17 m thick aquifer, confined at the bottom by a clay layer. Groundwater flow direction is south-southwest, determined by the recharge of the water from the Albert Canal flowing to the Kaatsbeek. Groundwater flow velocity and porosity are determined as 50 m $year^{-1}$ and 32% respectively. A groundwater investigation performed in 1992 indicated groundwater contamination with BTEX, which are organic solvents. BTEX is the acronym for benzene, toluene, ethyl benzene, and xylenes. The source of contamination is the leakage of underground solvent storage tanks. These leakages resulted in two contaminant source zones branching out in two 400 m to 500 m long plumes, crossing the border of the property of the car factory, as presented in Figure 3.1. The two adjacent contaminant source zones, merged into one. Average BTEX concentrations of about 238 mg L^{-1} were found at these merged source zones. The concentration diminished towards the end of the plume where concentrations of about 0.289 mg L^{-1} were observed.

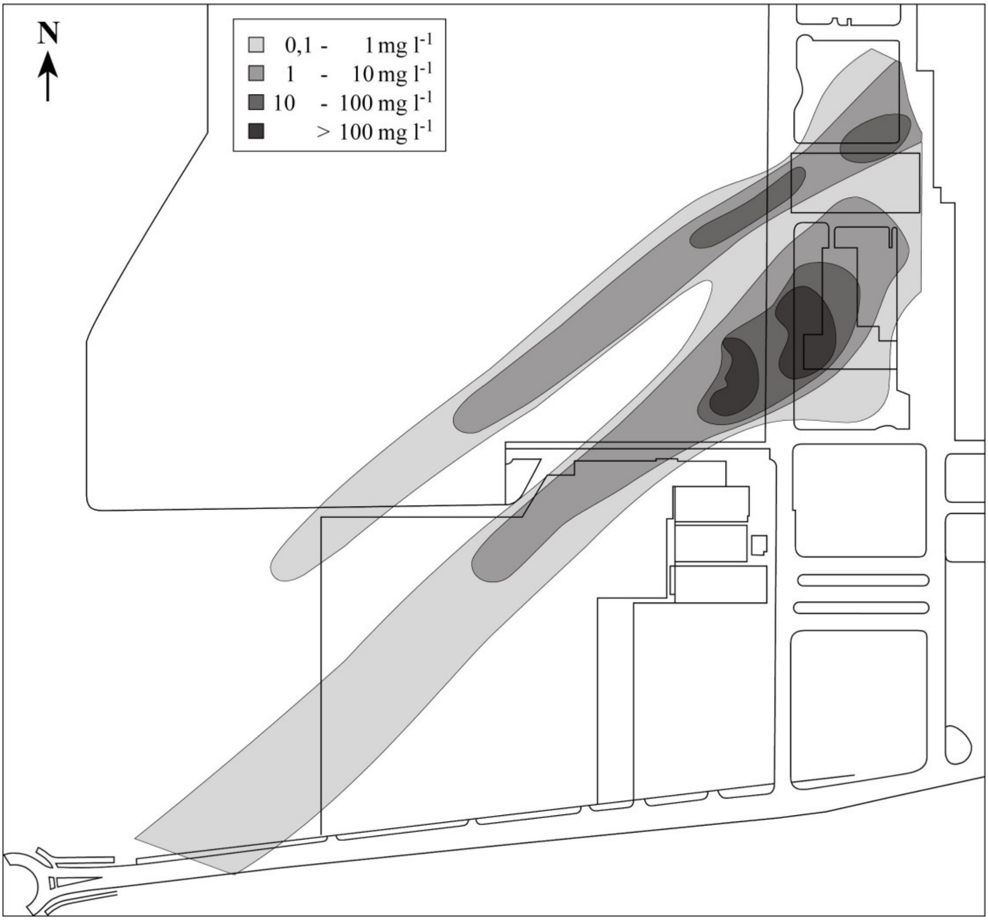


Figure 3.1 Plan view of the contaminated site

The responsible regulatory authority required the car company to contain the contamination within the borders of the property and set a remediation objective of 0.150 mg L^{-1} at the border of the site. Remediation activities started with removing the leaking tanks in 1995 and 1996. Three years later, a 10 500 m² area was planted with poplar trees 500 m from the source zone in rows perpendicularly on the groundwater flow direction. This plantation prevented the plumes from further expansion towards the Kaatsbeek [19]. In 2003, a P&T system was installed to remediate the source zone.

3.1.2 Simulation of groundwater flow and solute transport

The groundwater flow and transport model is developed using the code MOCDENS3D in which MODFLOW is integrated to calculate groundwater flow [58, 59]. This three-dimensional model is used to evaluate the effectiveness of phytoremediation compared to other, more conventional groundwater remediation technologies.

The finite-difference grid consists of 67 rows, 115 columns, and 17 layers. All cells of this grid have the same squared surface area with a side of 20 m and a height of 1 m. Two hydrogeological units are considered: a 7 m thick quaternary sand layer and a 14 m thick fine sand layer. The water table is situated 4 m below the ground surface. The model is calibrated to hydraulic heads and groundwater contaminant concentrations. Observed heads are primarily defined by the heads of the Albert Canal and the creek 'Kaatsbeek' which are the model boundaries. Therefore, the simulated heads fit well with the observed ones. Figure 3.2 presents the hydraulic heads and the location of the firm.

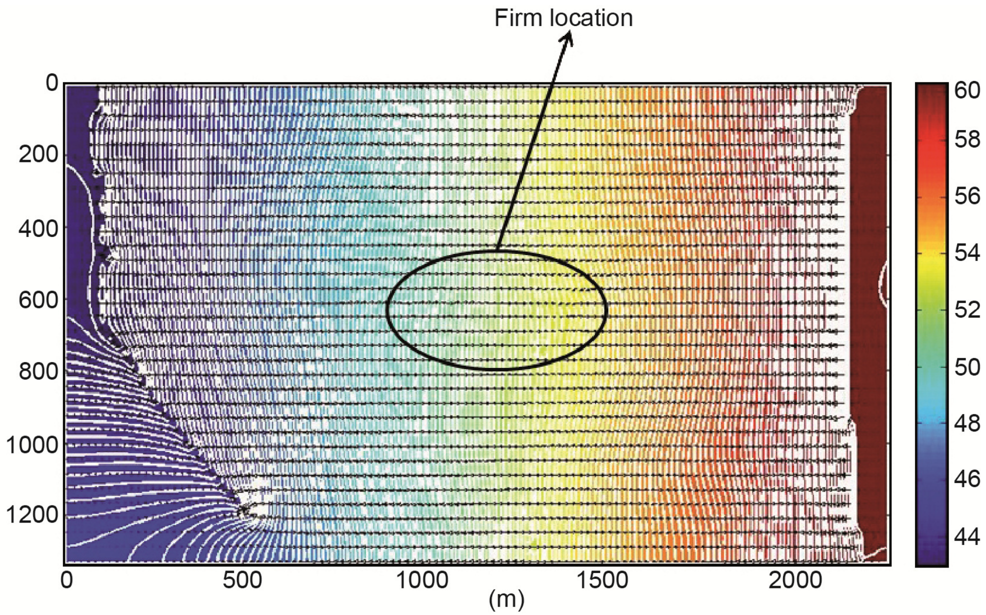


Figure 3.2 Representation of the hydraulic heads (varying between 43m and 60m) and the location of the firm

The applied horizontal conductivity is based on the results of a pumping test performed in 1998. All other parameter values [vertical conductivity (K_v), porosity (n), retardation factor (R), dispersion and degree of natural attenuation] are derived by the comparison of observed and simulated concentrations. K_v is deducted from the depth of the contaminant plume, n and R are estimated from the propagation characteristics (length, width and velocity) of the plume. Dispersion is determined by the width and depth of the transition zones. The degree of natural attenuation is based on the concentration gradients within the plume and is determined at 0.0026 d^{-1}. This value is within the range found by Kao and Wang [75] and Jeong *et al.* [76]. All the specified parameters are listed in Table 3.1. The effectiveness is indicated by the effect each remediation scenario has on the contamination. The effect is determined in (i) quantity of contaminant mass removed and (ii) remediation time.

Table 3.1 Hydrogeological properties

Hydrogeological parameter	Value
Horizontal conductivity (m d^{-1})	15.00 (Layer: 1 to 3); 1.00 (Layer: 4 to 17)
Vertical conductivity (m d^{-1})	0.3 (Layer 1-3); 0.037 (Layer 3-4); 0.02 (Layer 4-17)
Longitudinal dispersivity (m)	3.20
Horizontal transverse dispersivity (m)	0.15
Vertical transverse dispersivity (m)	0.09
Retardation factor (dimensionless)	2
Porosity (dimensionless)	0.32
Decay (d^{-1})	0.0026
Head Albert Canal (mTAW)[a]	60
Head Kaatsbeek (mTAW)	43

[a]mTAW is the second average leveling of Belgium. 0 mTAW = 2.3m below mean sea level

3.1.2.1 Simulation of contaminant occurrence

Before the simulation of each remediation technology, the occurrence of the contaminant source and plumes is simulated (from the first spill to the discovery of the contamination in 1992). Two sources are identified, covering surfaces of 1600 m² and 2400 m², respectively, from which a northern and southern plume branch out. In order to simulate the occurrence of the BTEX contamination, a contamination flux of 12 g m^{-2} d^{-1} during 20 years is introduced into the hydrogeological model. The simulated concentrations at the source zone and border of the site approach the measurements. Also the size of the modeled plumes matches well with the observations. Figure 3.3 shows the contamination dimensions.

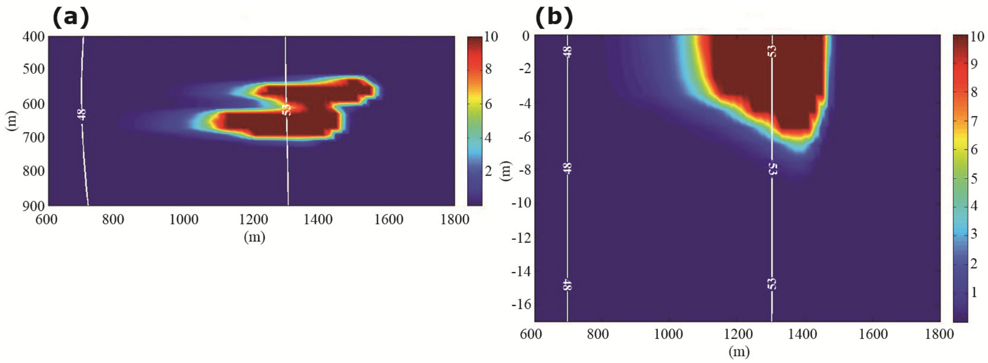


Figure 3.3 Horizontal (a) and vertical (b) cross section of the contamination. Concentrations are reported for a range between 0 and 10 mg L^{-1}

3.1.2.2 Simulation of remediation alternatives

After simulation of the evolvement of the plume and contaminant source, each remediation scenario is simulated. Phytoremediation is the remediation strategy that is actually being applied at this site. For this study, phytoremediation is compared to: (i) a P&T system, (ii) a vertical engineered barrier (VEB), (iii) a permeable reactive barrier (PRB), and (iv) natural attenuation (NA). As opposed to phytoremediation, these scenarios are hypothetical, except the P&T system at the source zone. Annex I briefly describes the different remediation strategies considered. The source removal and the P&T at the source zone is the same for each scenario. The plume remediation however changes in each scenario. Each remediation strategy is simulated from 1999 until 2018 but the operation time for each remediation option is different. The implementation of phytoremediation and the VEB involves mainly an investment cost and therefore these technologies are simulated for 20 years: from 1999 until 2018. Concerning P&T and the PRB, annual costs are high and therefore different models were run in order to determine the minimum required operation time. The PRB should operate for 3 years, the P&T system for 5 years. The results of the simulations indicate that when the PRB or the P&T are shut down earlier, the simulated concentrations would exceed the remediation objective.

P&T at the source zone and source weathering. The P&T system at the contaminant source zone started operating in 2003. The simulation of the P&T

system is based on the P&T operation time and the volume of water treated during that time. Groundwater is extracted from nine wells with screens in the eight uppermost layers at a rate of 7.42 m^3 d^{-1} for each extraction well. Although the leaking tanks have been removed, groundwater contamination continues from dissolution of a light non-aqueous phase liquid on top of the water table. Because the degree to which BTEX continued dissolving into the groundwater is not known, the source weathering rates were chosen so simulations matched observed concentrations. Until 2003, a source weathering rate of 12% $year^{-1}$ is applied. From 2003 on, when the P&T at the core is installed, a rate of 22% $year^{-1}$ is used. These rates fall within the range reported by Kampbell *et al.* [77] and Jeong *et al.* [76].

Because the remediation technologies have to prevent the contamination from further spreading, the effectiveness of each technology is determined at the area behind the location of the remediation technology *i.e.* 440 m and 540m from the core for the northern and southern plume respectively. Along each of these two lines, the maximum concentration at the end of each year is selected. This maximum concentration is an average concentration of eight cells with the same row and column from layer 1 to 8. Objective is to reach a maximum concentration below 0.150 mg L^{-1}.

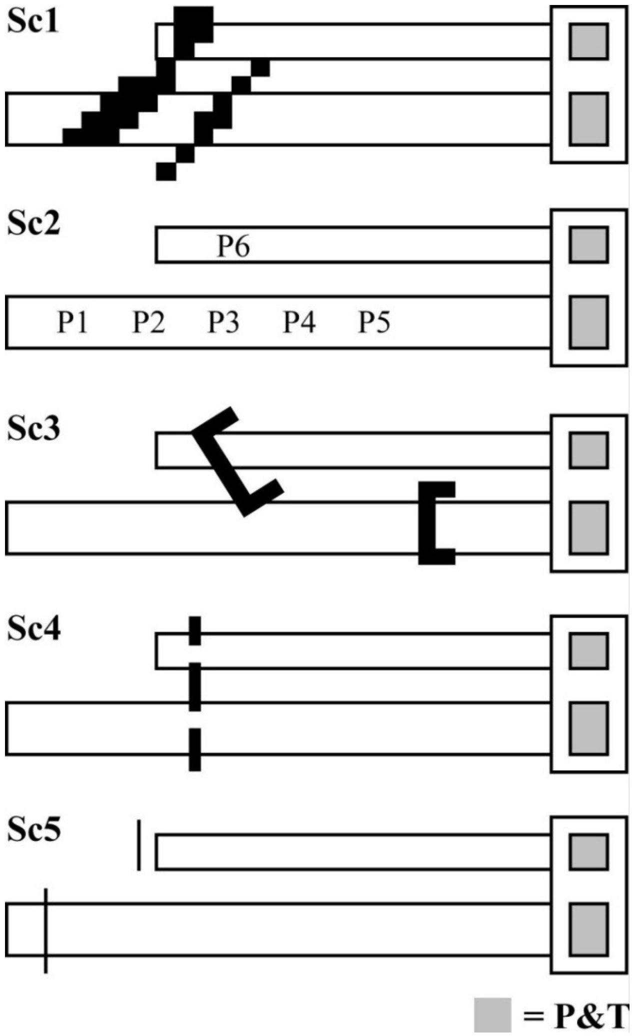


Figure 3.4 Plan view of a representation for the source zone and two plumes for five scenarios

(Sc1) Phytoremediation design with two plantations represented by black squares for 1999 to 2018.

(Sc2) Pump and treat design showing six extraction well locations for 1999 treat design showing six extraction well locations for 1999 to 2003.

(Sc3) Vertical engineered barrier design represented by two U-shaped walls for 1999 to 2018.

(Sc4) Permeable reactive barrier design shown by black rectangles for 1999 to 2001.

(Sc5) Natural attenuation for 1999 to 2018 showing two evaluation locations to determine the effect of each remediation technology.

Phytoremediation. Two hundred seventy five hybrid poplar trees (P. trichocarpa x deltoides cv. Hoogvorst and Hazendans) were planted in spring 1999 to remediate the BTEX plumes [19]. In 1999, concentrations up to 1 mg L^{-1} were observed under the plantation. From October 2002, the BTEX plume was stopped by the phytoremediation plantation and in June 2003, concentrations varied between 100 and 500 µg L^{-1}. Three years later, the BTEX plume had retreated from the planted area. Decrease of BTEX concentrations in the groundwater was not solely due to the high uptake capacity of the phreatophytic trees. When phytoremediation is applied, the microbial community in the rhizosphere increases in number and shows an increased uptake. Microorganisms can biodegrade a wide variety of organic contaminants [78], promote plant growth [12], and reduce evapotranspiration of volatile contaminants [20]. In this study, the presence of BTEX had a positive effect on the percentage of toluene degrading endophytic and rhizosphere bacteria associated with poplar trees. Toluene degrading bacteria can grow on toluene as a sole carbon source using the oxygen released into the soil by the poplar trees [19]. Poplars are flood – tolerant trees, able to maintain an aerobic environment in the rhizosphere. The trees absorb oxygen through the leaves and stems. Some of that oxygen is released from the roots into the rhizosphere [78].

Figure 3.4(Sc1) shows the 26 finite-difference cells in black representing the 10 400 m² planted surface. Evapotranspiration is simulated by specifying an extraction rate of 1.095 m³ d^{-1} per cell which is based on an evapotranspiration rate of 1 m $year^{-1}$ used at another site to simulate phytoremediation as well [79]. One finite – difference cell corresponds to eleven trees. Hence, the modeled mean evapotranspiration rate per tree equals 100 l d^{-1} which is in the range of reported ET rates by Barac *et al.* [19] (unpublished results).

Figure 3.5 shows the difference in hydraulic head with and without the poplar trees extracting groundwater. Due to a groundwater velocity of 50 m $year^{-1}$, groundwater extraction by trees does not lead to a depression cone.

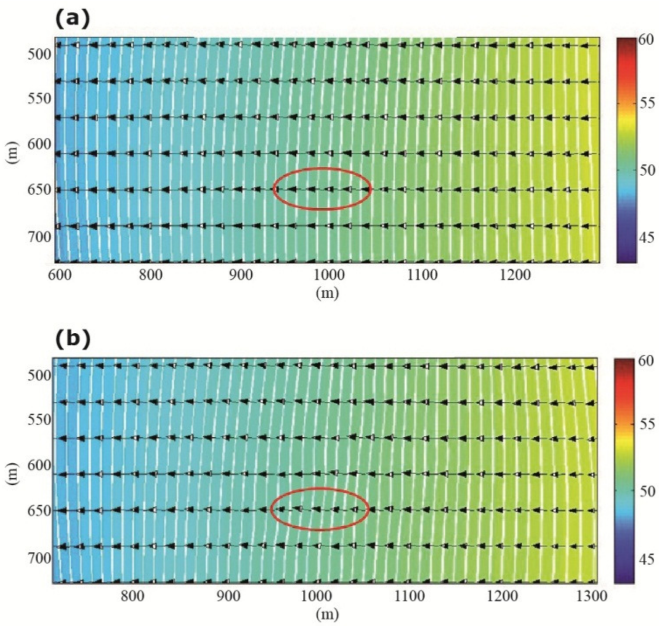


Figure 3.5 Difference in hydraulic head without (a) and with (b) groundwater extraction by poplar trees

Table 3.2 shows that when phytoremediation is simulated as a pump only, simulations do not match observed concentrations (See table 3.2, column 2). The contamination is not captured at the phytoremediation area. To simulate phytoremediation correctly, it is required to introduce a biodegradation rate in addition to the naturally occurring biodegradation rate of 0.0026 d^{-1}. These additional, self-selected biodegradation rates were deducted from the difference between the observed and simulated concentrations. At the start of the phytoremediation simulation, the mean biodegradation rate constant is 0.003 d^{-1} and increases in time. Increasing degradation is not surprising because a bacterial population needs time to adapt to the contamination and to acquire the specific degradation genes. Therefore, an efficient degradation level is simulated only after an acclimation period.

Table 3.2 Observed and simulated concentrations (mg L^{-1}) at the phytoremediation area. If biodegradation rates are introduced, simulated concentrations agree better with observed concentrations

For the year	Observed concentrations	Simulated concentrations at year end for phytoremediation	
		Biodegradation rate excluded	Biodegradation rate included
1999	0.413 [b]	0.595	0.438
2000	- [a]	0.636	0.447
2001	- [a]	0.645	0.446
2002	- [a]	0.618	0.422
2003	0.142 [b]	0.554	0.144
2004	0.015 [b]	0.481	0.016
2005	0.018 [b]	0.408	0.011
2006	0.004 [b]	0.318	0.002
2007	0.004 [b]	0.223	0.001
2008	0.001 [b]	0.147	0.001

[a] The quality of the data measured from 2000 to 2003 is highly uncertain and therefore this data was not used during the calibration.

[b] These are the results obtained from one well located in the center of phytoremediation area at the center line of the southern plume. The evolution of the concentrations measured at the other monitoring wells are not suitable to represent the effectiveness of phytoremediation because these wells are situated behind the phytoremediation area, towards the border of the site. At these wells contaminant concentrations below the remediation level are observed in 1999 already.

For this site, Barac *et al.* [19] observed that toluene degrading bacteria associated with poplar roots increased in number and by fraction of the microbial community in the presence of BTEX. An increase in the number of toluene degrading phenotypes was observed once the tree roots reached the BTEX plume, approximately 30 months after planting. Such a shift to toluene degrading strains was not observed for the bacterial communities associated with trees that grew outside the contaminated zone. This observation confirmed previous studies [16, 80-82]. In addition, horizontal gene transfer to adapt the endogenous microbial communities, as previously reported by Taghavi *et al.* [83], could have been involved and needed some time. Furthermore, in bacterial strains collected in 2006, when the concentrations of BTEX under the poplar trees had decreased below the detection limits, no bacteria capable of growing on toluene as a sole carbon source were found [19]. This illustrates the plasticity of the endogenous microbial communities in the presence of organic contaminants. In this study, the simulated biodegradation takes place only up until 4 m below the water table. The last column of Table 3.2 shows that when biodegradation is integrated, estimates match observed concentrations well. Figure 3.6 shows how the contamination is reduced after a phytoremediation operation period of 10 and 20 years.

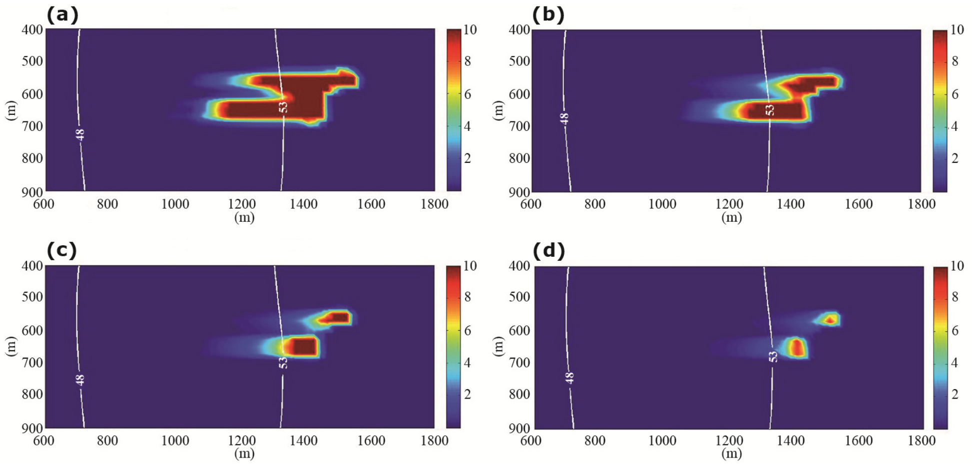


Figure 3.6 Horizontal cross section of the contamination concentrations (in mg L^{-1}) for a phytoremediation operation period of 5 years (a), 10 years (b), 15 years (c), and 20 (d) years

Pump & treat. The simulation of the P&T system for plume remediation involves five extraction wells (EW1 to EW5) drilled along the southern plume. Because the northern plume is partially situated under the factory building and in order to avoid indoor drilling costs, only one extraction well (EW6) is simulated at the end of the plume. Figure 3.7 shows the location of the simulated extraction wells.

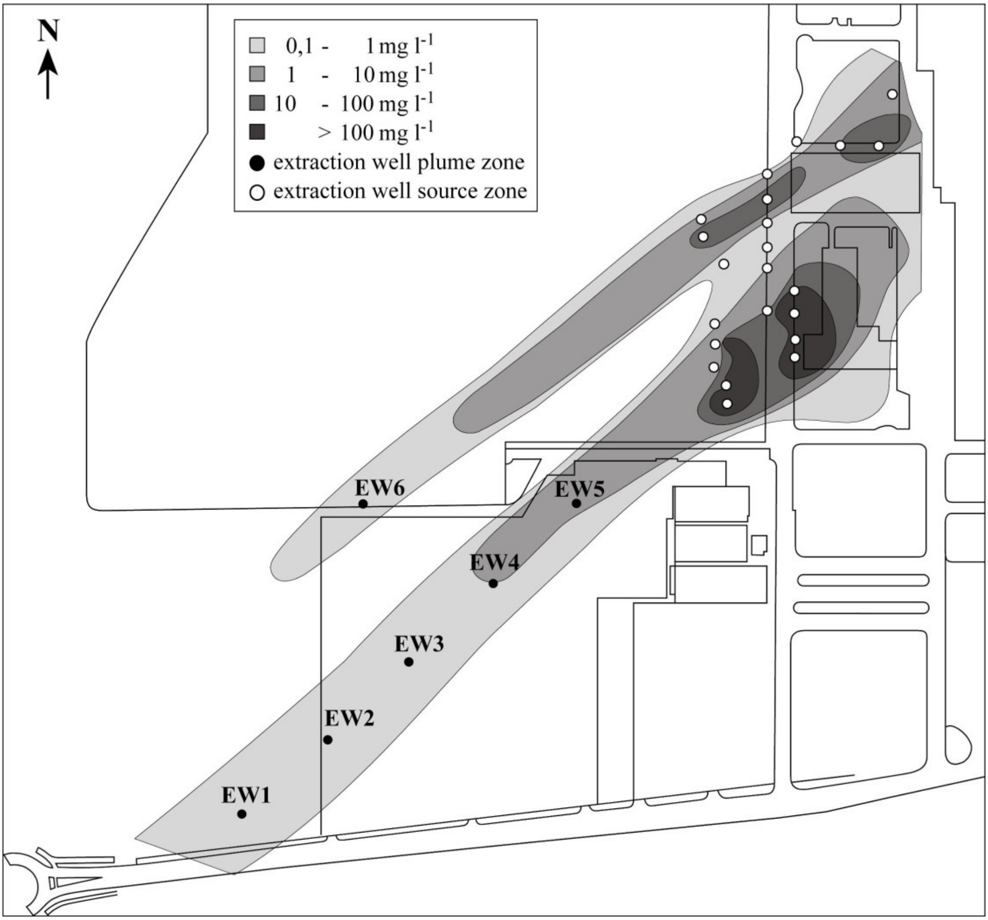


Figure 3.7 Design of the P&T remediation strategy

EW6, the extraction well at the northern plume is operating continuously at an extraction rate of 4 m³ h^{-1}. For the southern plume each year two pumps are operating, starting with EW1 and EW2. The extraction rate is 6 m³ h^{-1} for EW2, preventing any further contaminant transport down gradient. Well EW1, which is down gradient of well EW2, extracts contaminated groundwater at a rate of 3 m³ h^{-1}. In the second year, EW3 is used to capture the contaminated groundwater coming from the source zone and EW2 extracts the contaminated groundwater behind P3. In year three, EW4 captures the groundwater coming from the source zone, EW3 pumps the contaminated groundwater present behind EW4. In year four, EW5 captures the groundwater coming from the source, EW4 extracts the contaminated groundwater present behind EW5. In the fifth year, only EW5 operates at an extraction rate of 4 m^3 h^{-1}.

It is preconceived that the P&T system can only be shut down if at both the northern and southern plume, contaminant concentrations are below the remediation objective.

Figures 3.8a and 3.8b present the evolution of the simulated contaminant concentration during the different P&T operation periods, for the southern and northern plume respectively. In order to determine the minimum required operation period, P&T operation periods of two, four and five years are simulated. When Figure 3.8a is compared to Figure 3.8b, it is clear that the P&T system is more effective at the southern plume than at the northern plume. Although the objective at the southern plume can be reached during an operation period shorter than 5 years, the P&T system should continue operating in order to meet the objective at the northern plume. Figure 3.8b shows that the P&T system should stay operative during a period of 5 years. If the P&T system would be shut down earlier, the contaminant concentration at the northern plume would remain at a level that exceeds the clean up objective.

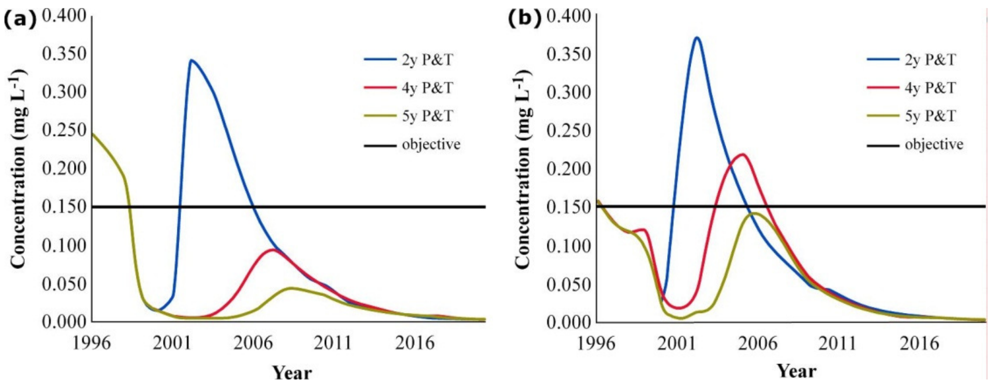


Figure 3.8 Evolution of contaminant concentration at the southern (a) and northern (b) plume for different operation periods of the P&T system

Figures 3.9 (a-e) show the hydraulic heads during the 5 years of operation. Concentrations are shown in a range of 0 to 10 mg L^{-1}. The pump & treat design at the contaminated plumes clearly results in a cone of depression, preventing the plumes from further spreading. Figure 3.9 (f) presents a vertical cross section of the contamination and the velocity vectors, indicating groundwater flow velocity and direction. The larger the vector, the stronger the groundwater flow.

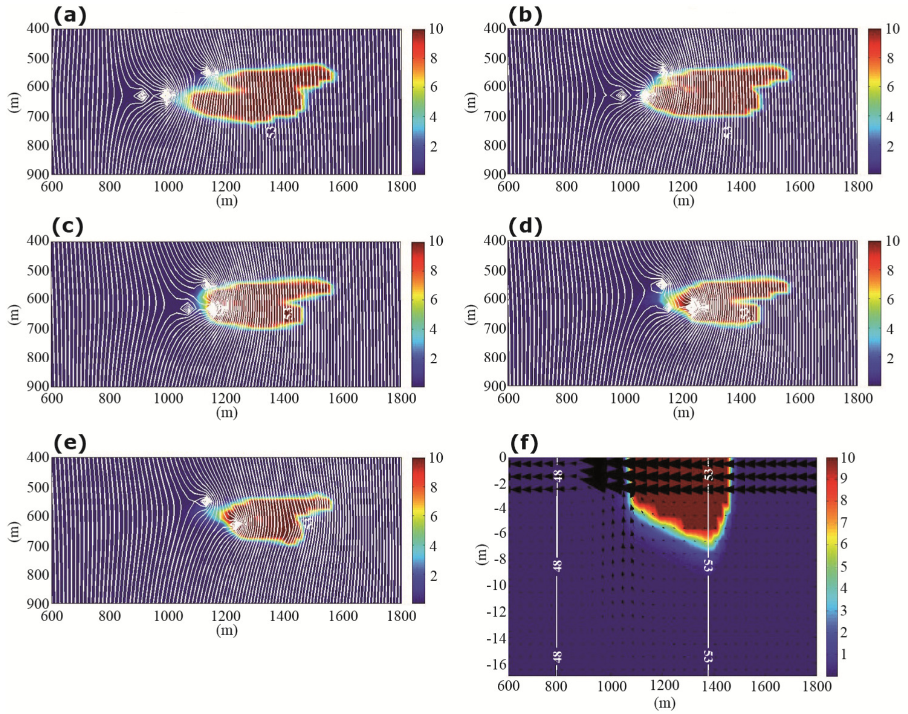


Figure 3.9 Horizontal cross section of the contamination and hydraulic heads after 1 year (a), 2 years (b), 3 years (c), 4 years (d) and 5 years (e) of operation and a vertical cross section of the contamination with an indication of the velocity vectors (f)

Vertical engineered barrier (VEB). For the simulation of the VEB, the barriers are put as close as possible to the core (Figure 3.4 (Sc3)). As the northern contaminant plume flows under the factory building, the barrier could not be placed perpendicular to the plume nor closer to the core. In order to simulate the low permeability, the horizontal conductivity is specified 25 times lower for the cells representing the VEBs. The barriers reach a depth of 6 m below the water table. Figures 3.10 a, b, c, and d present the evolution of the contaminated area after 5, 10, 15 and 20 year respectively. Concentrations are reported in a range of 0 to 10 mg L^{-1}. It is shown that concentrations decrease during the remediation period.

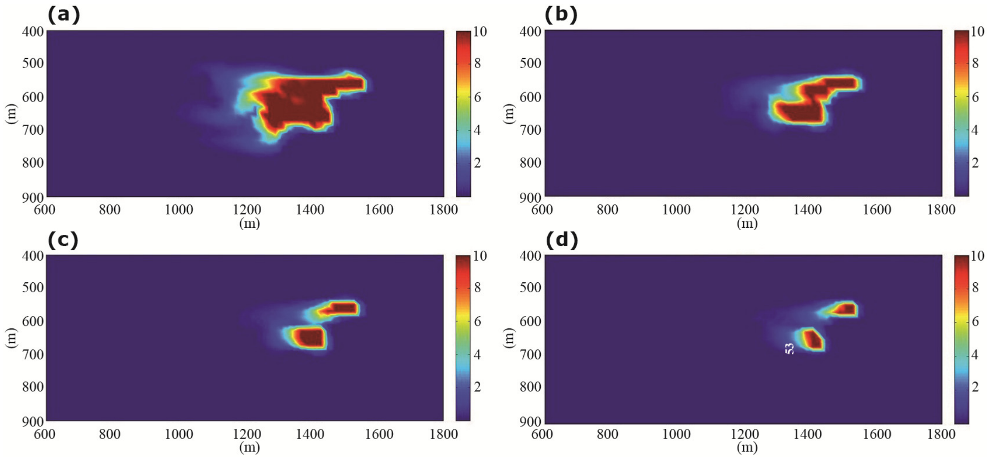


Figure 3.10 Horizontal cross section of the contamination for hydraulic barriers after 5 years (a), 10 years (b), 15 years (c), and 20 years (d) of operation

Figure 3.11 (a) presents a horizontal cross section that indicates the hydraulic heads (from 43 to 60 mTAW) and the velocity vectors. This figure shows that groundwater flows along the hydraulic barriers. Nevertheless, Figures 3.10 (a-d) show that the quantity of contamination passing the barriers is limited. A vertical cross section of the contamination (concentration ranging from 0 to 10 mg L^{-1}) is presented in Figure 3.11 (b). It shows that although groundwater flows below the hydraulic barrier, the contamination does not pass the barrier from underneath.

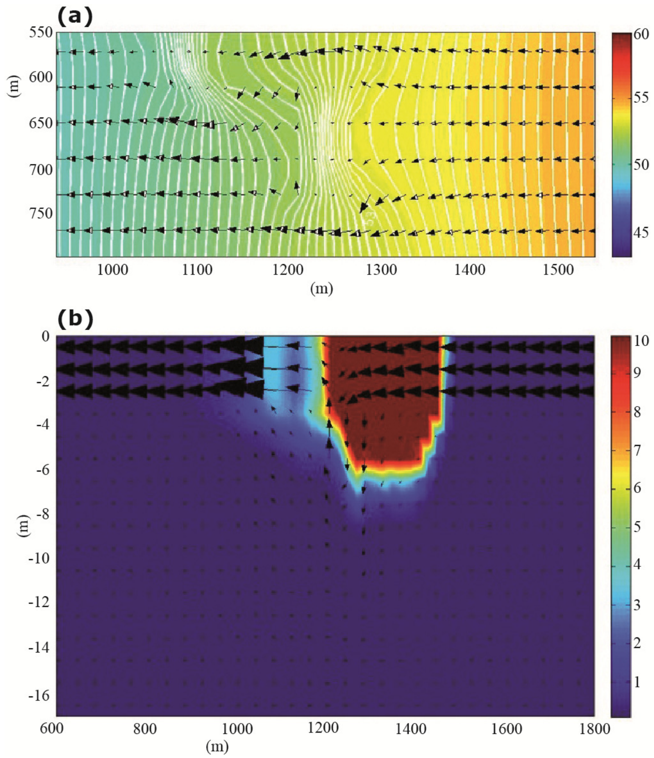


Figure 3.11 Horizontal (a) and vertical cross (b) section of the site presenting the velocity vectors for the hydraulic barriers. The horizontal cross section presents the hydraulic heads (in mTAW), the vertical cross section presents the contamination (in mg L^{-1})

Permeable reactive barrier. A continuous trench system is simulated like presented in Figure 3.4(Sc4). For the trench area, a biosparging system is assumed. Biosparging is similar to air sparging which is a frequently applied *in* *situ* remediation technology for the remediation of BTEX contamination [5, 84, 85]. Biosparging involves the injection of air into the saturated zone at a low rate in order to stimulate aerobic biodegradation. By comparison, air sparging injects air at a much higher rate, stimulating volatilization of the contamination and then, soil vapour extraction is required to treat the BTEX enriched gas phase [84-86]. Bowles *et al.* [84] report a decrease of total BTEX concentrations from 0.183 mg L^{-1} (when entering the PRB) to 0.001 mg L^{-1} (before leaving the PRB). Based on the results reported by Bowles *et al.* [84], the biosparging system is simulated with a degradation rate of 0.095 d^{-1}.

Figure 3.12a and 3.12b present the evolution of the simulated contaminant concentration during different PRB operation periods, for the southern and northern plume respectively. Operation periods of two, three and four years are simulated. Unlike the P&T system, the PRB is more effective in remediating the northern plume than the southern plume. Figure 3.12a shows that the PRB should stay operative during a period of 3 years.

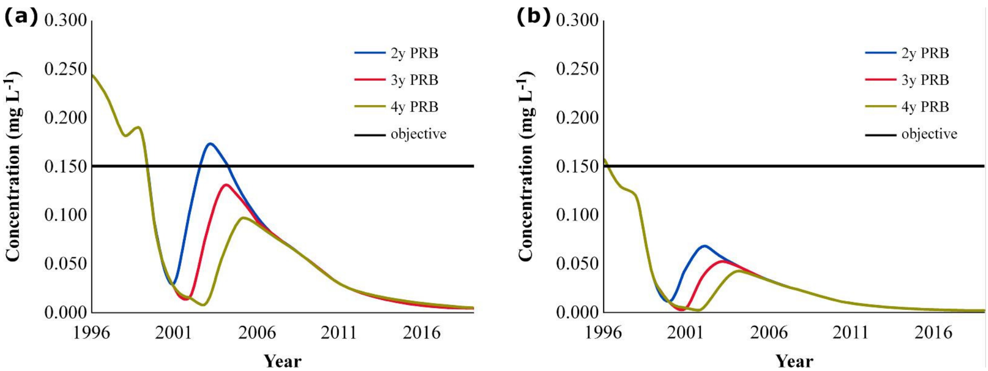


Figure 3.12 Evolution of contaminant concentration at the southern (a) and northern (b) plume for different remediation periods of the PRB system

Figures 3.13 (a-d) present the horizontal cross section of the contamination (concentrations range between 0 and 10 mg L^{-1}) for 1, 5, 10, and 20 years operation period.

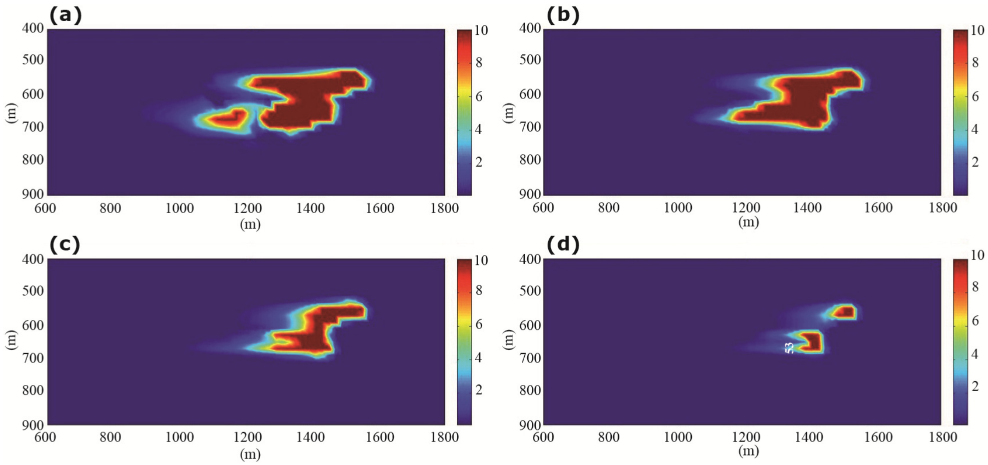


Figure 3.13 Horizontal cross section of the contaminated site for the permeable reactive barrier indicating contaminant concentration (in mg L-1) after a 1 year (a), 5 years (b), 10 years (c) and 20 years (d) operation period

Figures 3.14 (a-b) present the velocity vectors. Figure 3.14 (a) presents a horizontal cross section indicating the hydraulic heads. It is shown that the groundwater flows through the gate. Figure 3.14 (b) presents a vertical cross section of the contaminant concentrations (mg L^{-1}). The contamination does not flow underneath the barrier.

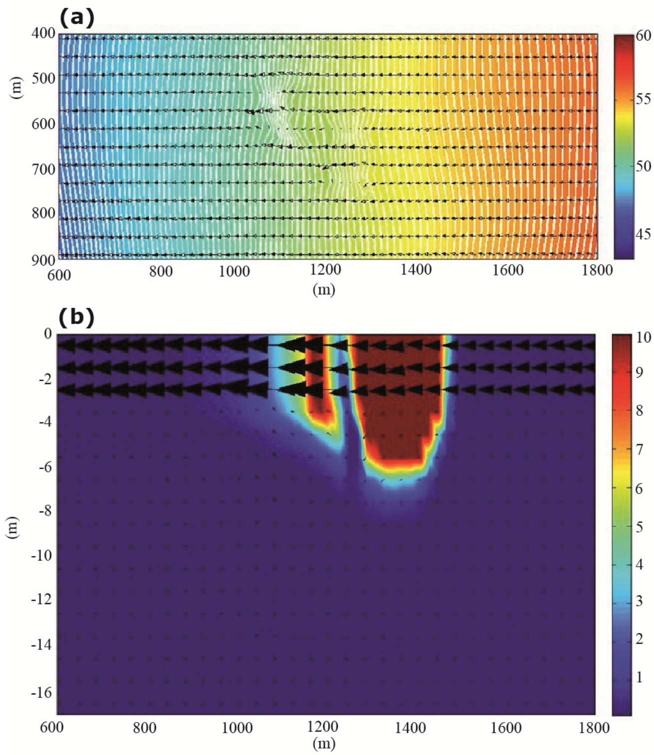


Figure 3.14 Horizontal (a) and vertical cross (b) section of site presenting the velocity vectors for the permeable reactive barrier. The horizontal cross section shows the hydraulic heads (in mTAW), the vertical cross section presents the contamination (in mg L^{-1})

Natural attenuation (NA). In this case, no measures are taken to remediate the contaminant plumes and remediation is left to natural attenuation (Figure 3.4(Sc5)). The simulation of the source weathering and the operation of the P&T system at the core stay the same. Figure 3.15 presents the horizontal cross section, indicating how the contamination has evolved after 10 and 20 years.

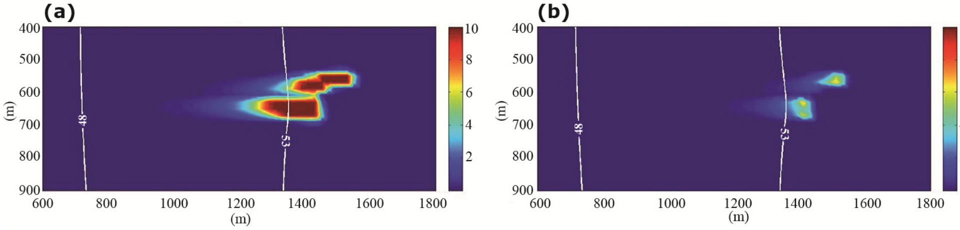


Figure 3.15 Horizontal cross section for the natural attenuation strategy indicating the contaminant concentration (mg L-1) after a 10 years (a) and 20 years (b) operation period

3.1.3 Results of the ACERs and ICERs: a phytoremediation case study

3.1.3.1 Costs and effects

The effect is determined down gradient of each remediation zone, *i.e*. 440 m and 540 m from the core for the northern plume and southern plume, respectively. The effect is determined in (i) time required to reach this objective (remediation time) and in (ii) mass removed during that period. Figure 3.4(Sc5) shows the location where the effect is determined. Table 3.3 shows the results. The remediation objective is considered to be reached when the objective is met at both the northern and southern plume. Phytoremediation, VEB and natural attenuation are simulated to stay operative during 20 years. The PRB and P&T system stay operative in the simulation as long as the remediation objective is not met at one of both plumes. Note that the model results are produced for each discrete year. Table 3.3 shows that when P&T and phytoremediation are applied, in both cases the objective is already reached after one year. However, during that year (1999) the P&T system removed more contaminant mass. When the VEB and PRB are compared, a similar conclusion can be drawn. While the remediation period is 2 years for both remediation strategies, when the PRB is applied, more contaminant mass is removed during those 2 years. Also the operation time of each technology and the mass removed or contained during the operation time are presented in Table 3.3 because the remediation technologies need to remain operative after the remediation objective is reached in order to minimize rebound effects. When PRB or P&T are adopted, most mass is removed during the short operation time, while in case of NA, phytoremediation and VEB, a longer period is needed to remove a similar quantity of contaminants. Figure 3.16 shows the evolution of the concentration for the different remediation strategies graphically.

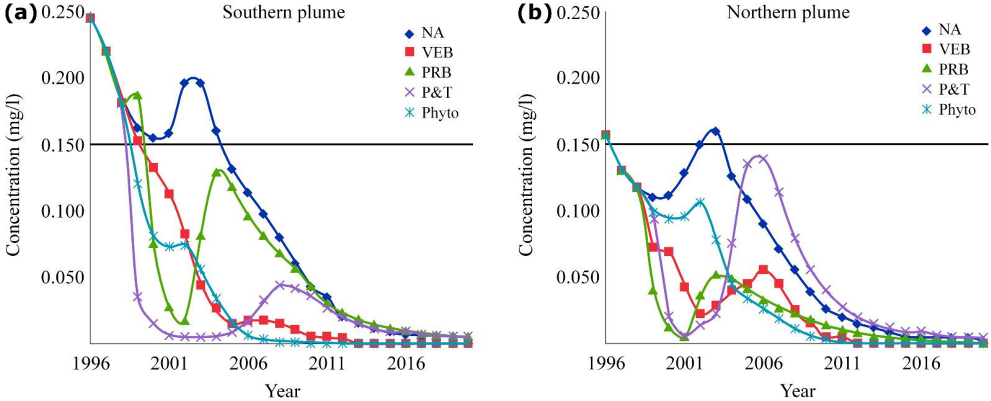


Figure 3.16 Evolution of contaminant concentrations

Total discounted remediation costs are presented in the last row of Table 3.3. Remediation costs of the PRB and P&T system are calculated throughout a period of three and five years, respectively, being the intended operation time of these remediation technologies. For phytoremediation and VEB the major investment costs are made in the first year of operation. For phytoremediation only some mowing costs are considered for the years following the plantation. Total costs are discounted to 1999, using a discount rate of 5%. Monitoring costs are assumed to be the same, irrespective of remediation technology used. Total discounted monitoring costs are determined for a period of 20 years and equal €54 610.

Table 3.3 Mass removal and total discounted costs for remediation alternatives (all start in 1999)

	Phyto	P&T	VEB	PRB	NA
Year (end) objective is reached (SP)	1999	1999	2000	2000	2005
Remediation period (years)[a]	1	1	2	2	7
Mass removed from both plumes during remediation period (g)[b]	161	345	200	434	120
Operation period (years)	20	5	20	3	20
Mass removed from both plumes during operation period (g)[c]	612	556	612	546	591
Total discounted costs (1000€)	97	153	341	155	55

[a]It is assumed that the remediation objective is met when concentrations at both plumes are lower than 0.150 mg L^{-1}. Because it takes longer to achieve the objective at the southern plume (SP) than at the northern plume (NP), the remediation time equals the time necessary to reach the objective at the SP. After the source removal in 1996, concentrations at the end of the plumes start decreasing immediately due to natural attenuation. For the northern plume, the remediation objective is already met in 1998 (not shown).

[b]To calculate mass removed, first the mass present is determined for each year. The maximum concentrations (not shown) are multiplied with the porosity, the retardation factor and a groundwater volume of 20 m*20 m*8 m. The maximum concentration is an average concentration of eight cells with the same row and column from layer 1 to 8. Mass removed is the difference between the mass in two successive years.

[c]Mass removed during the time of operation is calculated using the simulated concentrations at the end of the time of operation (not shown).

3.1.3.2 Average cost effectiveness analysis (ACER)

The ACER is first calculated in cost per year of remediation time and in cost per mass removed or contained during the remediation time. According to this analysis, respectively NA and PRB should be the preferred remediation options because these alternatives are attended with the least average cost (Table 3.4, indicated in bold). However, ‘remediation time’ does not seem to be a suitable parameter to determine the average cost. The P&T system and PRB continue operating after the remediation objective has been reached to minimize the increase of concentration after the system is stopped and therefore ‘operation time’ is preferred to determine the average annual cost. When the cost per year operation time is considered to calculate ACER, the order of preferred remediation option is as follows: NA – Phytoremediation – VEB – P&T – PRB. When the ACER is determined in mass removed during the operation time, the order of preference changes: NA – Phytoremediation – P&T – PRB – VEB.

For all the remediation strategies, the revenues from selling the property after the remediation objective is reached, are not taken into account. When natural attenuation is applied, the remediation objective is reached at the lowest cost. However, when the other remediation strategies are applied, the remediation objective is reached earlier and the property can be sold and redeveloped sooner. Hence, considering natural attenuation, the firm loses revenue that could have been obtained by applying another remediation strategy. This opportunity cost is not taken into account. In Chapter 5, revenues resulting from the property sale are introduced in the economic model.

Table 3.4 Calculation of the ACER and ICER

ACER					
Remediation strategy	Phyto	P&T	VEB	PRB	NA
Period considered: remediation time					
Cost per year (€)	96 621	153 146	170 383	77 580	**7802**
Cost/mass (€ g^{-1})	599	443	1 707	**358**	454
Period considered: operation time					
Cost per year (€)	4 831	30 629	17 038	51 720	**2 731**
Cost/mass (€ g^{-1})	158	276	557	284	**92**

ICER				
Remediation strategies[a]	P&T	VEB	PRB	NA
Period considered: remediation time				
ΔC	56 525	244 145	58 540	-42 010
ΔE: mass removed (g)	184	38	272	-41
ICER (€ g^{-1})	307	6 358	215	1026
ΔE: time (year)	0	-1	-1	-6
ICER (€ $year^{-1}$)	/	-244 145	-58 540	7 002

[a]Phytoremediation is assumed to be the reference remediation strategy to which the alternative strategies are compared.

3.1.3.3 Incremental cost effectiveness analysis (ICER)

To determine the incremental cost effectiveness ratio (ICER), a trade-off between extra costs and extra effects (*i.e.* faster clean up or more mass removed or contained) is calculated and therefore, remediation time is a suitable quantification of the effect. In this case, the extra cost of an extra gram of contaminants removed or contained and the extra cost to reach the remediation objective of 0.150 mg L^{-1} one year sooner is examined. Phytoremediation is the reference to which the other remediation options are compared. The lower part of Table 3.4 gives an overview of how the ICERs are determined when the effectiveness is measured in remediation time and mass removed or contained during the remediation time. Figure 3.17 shows that the use of the ICER does not lead to a definitive choice of remediation technology. When the effect is calculated in mass removed or contained (Figure 3.17a), the installation of a VEB, a P&T system or a PRB is more effective, but more expensive than phytoremediation. Natural attenuation is less effective but less expensive than phytoremediation. Only the VEB can be ruled out because the PRB and P&T are more effective and less expensive than the VEB.

When the effect is measured in remediation time (Figure 3.17b), phytoremediation dominates the VEB, the PRB and the P&T system. In terms of remediation time, the NA alternative is ambiguous compared to phytoremediation. Therefore, in both terms of mass removal and remediation time, the decision maker should consider to progress to a cost benefit analysis and attach a value λ to the incremental effect (*i.e.* faster clean up or more mass removed or contained).

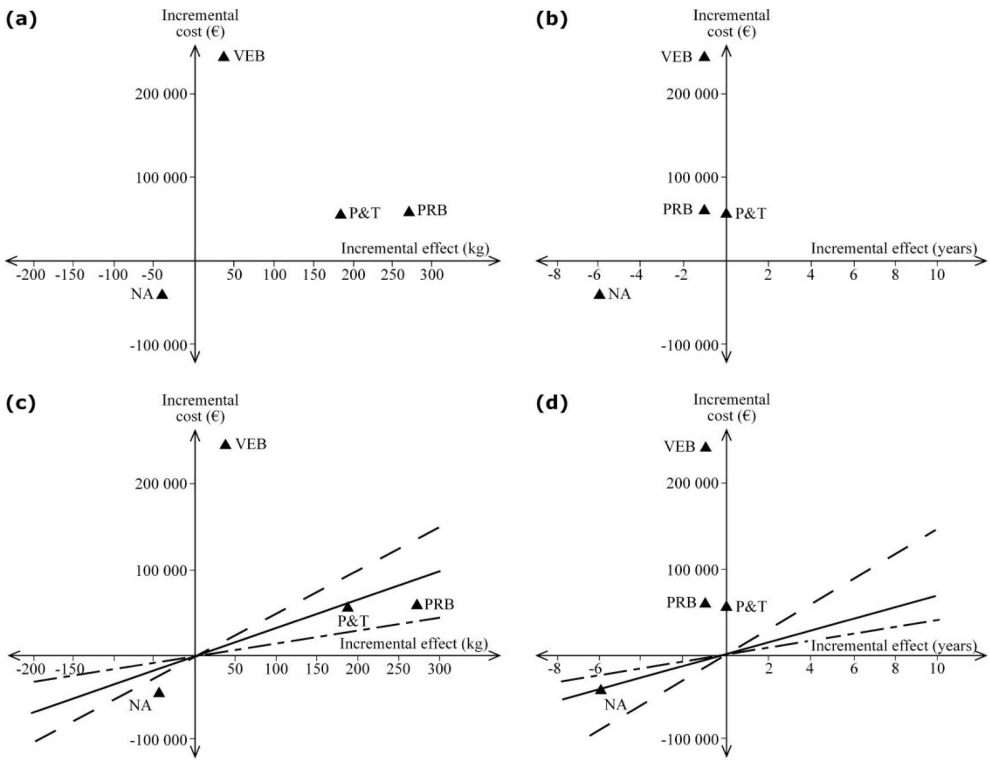


Figure 3.17 Cost effectiveness plane. determination of the ICER and valuation of the incremental effect (value of removing an extra gram of contamination and value of reaching the remediation objective one year earlier)

(a) Representation of the ICER in € g^{-1}

(b) Representation of the ICER in € $year^{-1}$

(c) Valuation of removing an extra gram contamination: the solid line represents λ = €321 g^{-1}, the dashed line represents λ = €500 g^{-1}, the dash-dotted line represents λ = €150 g^{-1}.

(d) Valuation of reaching the objective one year earlier: the solid line represents λ = €7002 $year^{-1}$, the dashed line represents λ = €15 000 $year^{-1}$, the dash-dotted line represents λ = €4 000 $year^{-1}$.

The value *λ* represents the willingness to pay for an incremental effect. Using this value, the net benefit of applying another technology than the reference technology, can be calculated. Table 3.5 shows the calculated net benefit for different values of *λ*. For each value of *λ*, the remediation scenario with the highest net benefit (indicated in bold) is preferred. When the effect is calculated in mass removed, no matter the value of *λ*, NA or the PRB will always have a net benefit greater or equal to zero when compared to phytoremediation. The net benefits of the PRB and NA are equal when λ, the value of an extra gram contaminant mass removed, is determined at €320. When λ is lower than €320 g^{-1} (*e.g.,* €150 g^{-1}), NA should be chosen because NA has the highest net benefit. When λ is higher than €320 (*e.g.,* €500 g^{-1}), the PRB has the highest net benefit, and hence is preferred.

Figure 3.17c presents this graphically. The solid line represents the willingness to pay when λ is equal to €320 g^{-1}. The distance between the ICER of the PRB and the solid willingness-to-pay-line is equal to the distance between the ICER of NA and the solid willingness-to-pay-line. When λ is higher than €320 g^{-1}, both the ICERs of NA and the PRB are below the willingness-to-pay-line. This is the dashed line in Figure 3.17c. The PRB is the preferred technology because the distance between the ICER of the PRB and the dashed line is greater compared with the ICER of NA. When λ is lower than €320 g^{-1}, represented by the dash-dotted line, only the ICER of the NA is situated below the willingness-to-pay-line. Hence, NA is preferred.

Table 3.5 Net benefit (NB) in € for different λ (€ g^{-1} and € $year^{-1}$)

	Mass removed			Remediation time		
Strategy	λ = 320	λ = 500	λ = 150	λ = 7 000	λ = 15 000	λ = 4 000
P&T	2 622	35 635	-28 877	-56 525	-56 525	-56 525
VEB	-231 823	-224 945	-238 385	-251 147	-259 145	-248 145
PRB	**28 867**	**77 652**	-17 682	-65 542	-73 540	-62 540
NA	**28 867**	21 530	**35 866**	**0**	-47 990	**18 010**

When the effect is measured in remediation time, phytoremediation is the preferred strategy if the value of reaching the remediation objective one year sooner is higher than €7 000. In that case, the net benefit of all other remediation strategies is negative (Table 3.5) and the ICERs are above the willingness-to-pay-line (Figure 3.17d). The solid line is the willingness-to-pay-line when λ is equal to €7 000 $year^{-1}$, the dashed line represents the willingness-to-pay when λ is greater than €7 000 $year^{-1}$. If λ is smaller than €7 000 $year^{-1}$, then the net benefit of NA is positive and hence NA is preferred. The dash-dotted line represents the willingness-to-pay when λ equals €4 000 $year^{-1}$.

In the further analyses, the number of remediation alternatives is reduced. Phytoremediation is only compared to natural attenuation and pump & treat. These remediation strategies are mostly applied for groundwater remediation. The permeable reactive barrier and the vertical engineered barrier are not taken into consideration anymore. The previous analysis makes clear that the vertical engineered barrier is not an alternative. Although the permeable reactive barrier seems to be an effective alternative, it should be noted that the biodegradation rate applied in the simulation, is based on literature. It is not proven that this remediation strategy would be effective at the site under consideration. The impact of technological uncertainty will be discussed and empirically applied in Chapter 4 and Chapter 5.

3.1.4 The value of groundwater remediation: cost of illness

In this section, the results of the hydrogeological model are used as input values for a health impact assessment. Based on the risk of adverse health effects due to contaminant exposure, the cost of illness is calculated to value groundwater remediation for the site under consideration. BTEX (benzene, toluene, ethylbenzene and xylene) are aromatic hydrocarbons. Benzene is classified as a carcinogenic contaminant that can cause leukaemia after a lifetime exposure to low concentrations of benzene [87]. TEX are non-carcinogenic but can cause irritation of the bronchial tubes, dizziness, and kidney damage after exposure [88-90].

To assess the health effects of the contamination, the software program Vlier Humaan, developed by the company Van Hall Larenstein, is used. Based on the contaminant concentration in groundwater, this software program calculates the quantity of daily intake. This quantity is then compared to the permissible level of daily intake. When the effect of non-carcinogenic contaminants is assessed, the population that experiences the adverse health effects is not quantified. Whereas for carcinogenic contaminants, the risk index calculated by the model indicates the increase in leukaemia cases for 100 000 persons who are exposed to the contaminant during a lifetime. Because the risk of a carcinogenic contaminant is better quantifiable, the health effects are only assessed for benzene. The risk index is calculated for the concentration simulated at the end of the southern plume for the phytoremediation, pump & treat, and natural attenuation strategies. The results are presented in Table 3.6. For each year, the calculated risk index is smaller than 1, which means that the risk of adverse health effect is limited. For year 0, considering the employees of the company as the population exposed to the contamination, an increase of 0.00155 cancer cases is calculated.

Table 3.6 Risk index calculated by Vlier Humaan for different remediation strategies

Remediation strategy	NA		P&T		Phyto	
	Conc[a]	RI	Conc[a]	RI	Conc[a]	RI
Year 0	0.268	1.6E-3	0.268	1.6E-3	0.268	1.6E-3
Year 5	0.184	8.3E-4	0.013	5.9E-5	0.097	4.4E-4
Year 10	0.121	3.2E-4	0.082	2.2E-4	0.012	3.2E-5
Year 15	0.027	5.7E-5	0.038	8.0E-5	0.001	2.1E-6
Year 20	0.006	1.3E-5	0.008	1.7E-5	-	-

[a]Concentration in mg L^{-1}

The avoided health costs are considered as the benefit of the remediation. The net benefit of each groundwater remediation strategy leads to the avoidance of adverse health effects. To value the cost of adverse health effects, the extra cases of leukaemia are quantified in disability adjust life years (DALY) and a monetary value is attached to each DALY. To calculate the benefit of the remediation, the difference in health costs is calculated for different periods. Torfs [91] estimates that one case of leukaemia corresponds to 16.3 DALY and this author values one DALY at €78 501 for the Flanders region. Using these figures, Table 3.7 presents an overview of the estimated DALY for the different remediation strategies and after different remediation periods. The avoided quantity of DALY is calculated with reference to year 0.

Table 3.7 Calculation of the avoided DALY for each remediation strategy

strategy	NA		P&T		Phyto	
	DALY	Avoided DALY	DALY	Avoided DALY	DALY	Avoided DALY
Year 0	0.0253		0.0253		0.0253	
Year 5	0.0008	0.0118	0.0010	0.0243	0.0071	0.0182
Year 10	0.0003	0.0201	0.0035	0.0217	0.0005	0.0247
Year 15	0.0001	0.0243	0.0013	0.0230	0.0000	0.0252
Year 20	0.0000	0.0251	0.0003	0.0250	-	0.0253

The calculation of the net benefit for each remediation strategy for different remediation periods is shown in Table 3.8. These results show that the avoided health costs are limited, which results in a negative net benefit. When natural attenuation is applied, the net benefit is the highest (See Table 3.8, indicated in bold). Based on these calculations, natural attenuation is the preferred remediation strategy.

Table 3.8 Calculation of the net benefit (NB in 1000€)for each remediation strategy

	NA			P&T			Phyto		
Year	Benefit	Cost	NB	Benefit	Cost	NB	Benefit	Cost	NB
5	0.9	36.7	**-35.8**	1.9	135.4	-133.5	1.4	72.9	-71.5
10	1.6	44.6	**-43.1**	1.7	143.3	-141.6	1.9	86.7	-84.7
15	1.9	50.7	**-48.8**	1.9	149.4	-147.5	2.0	92.7	-90.7
20	2.0	54.6	**-52.6**	2.0	153.3	-151.3	2.0	96.6	-94.6

The calculation of the ACER in section 3.1.3.2 already showed that for the site considered, natural attenuation seems to be a more cost effective solution to the groundwater pollution than phytoremediation. The ICER presented in section 3.1.3.3 led to a more nuanced conclusion, indicating that in order to make a decision, a value should be attached to the incremental effect. If this value is determined in terms of avoided health costs, natural attenuation is the preferred remediation strategy. However, the cost of illness is only related to the health risk associated with the contaminated site. Also the risk of further spreading of the contamination should be taken into account when evaluating the benefits of groundwater remediation. Without the phytoremediation area, the contamination may pollute adjacent sites and reach the creek 'Kaatsbeek'. This topic is not further considered in this dissertation and is subject to future study.

3.1.5 Change in hydraulic gradient

A specific characteristic of the BTEX contaminated site considered, is the hydraulic gradient of 0.008 which induces an increased groundwater flow. In this section, it is studied whether the selected groundwater remediation strategy changes when the hydraulic gradient is decreased by half, to 0.004. The remediation strategies considered are: phytoremediation, pump & treat, and natural attenuation.

3.1.5.1 Simulation of groundwater flow and solute transport

The occurrence of the contamination is simulated like presented in paragraph 3.1.2.1. Two sources are identified, covering surfaces of 1600 m² and 2400 m², respectively, from which a northern and southern plume branch out. A contamination flux of 12 g m^{-2} d^{-1} during 20 years is introduced into the hydrogeological model. Figure 3.18 shows the concentrations in the first layer of the groundwater model after the simulation has run for 20 years for a gradient of 0.008 (a) and 0.004 (b). When a hydraulic gradient of 0.004 is applied, the groundwater flow is reduced and hence, the contamination is spread to a smaller extent (See Figure 3.18b).

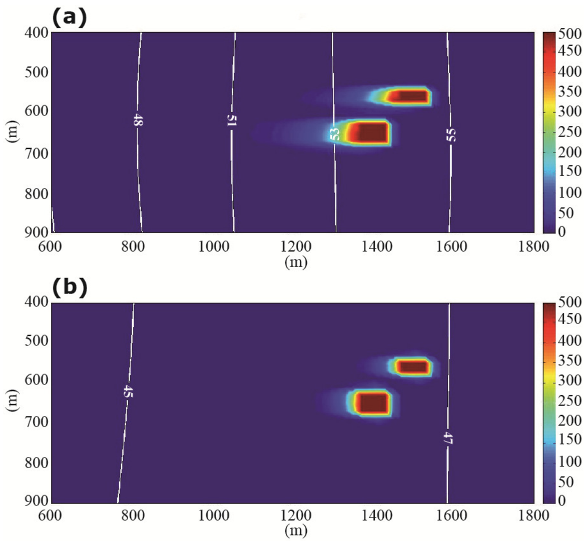


Figure 3.18 Spread of contamination in the first layer of the hydrogeological model

3.1.5.2 Design of the groundwater remediation strategies

For this case, the timing of the different remediation strategies at both the source and plume zones is the same as explained in paragraph 3.1.2.2. Also the operation of the P&T at the source zone does not change. To prevent the plumes from further spreading, three remediation strategies are considered: natural attenuation, phytoremediation, and P&T. The phytoremediation strategy is simulated as explained in paragraph 3.1.2.2. Only the area where the trees are located is changed, the phytoremediation area is situated closer to the source zone (see Figure 3.19).

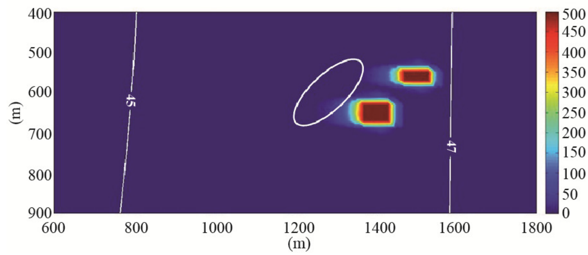


Figure 3.19 Location of the phytoremediation area

Regarding the simulation of the P&T remediation strategy, one extraction well is located at the end of the northern plume, three extraction wells are operating at the southern plume. Groundwater will be extracted in a similar manner like explained in paragraph 3.1.2.2. From the extraction well closest to the source zone groundwater is extracted at 4 m^3 h^{-1}. An extraction rate of 2 m^3 h^{-1} is applied for the extraction well behind. Figure 3.20 presents the evolution of the concentration and the configuration of the extraction wells for the P&T remediation strategy.

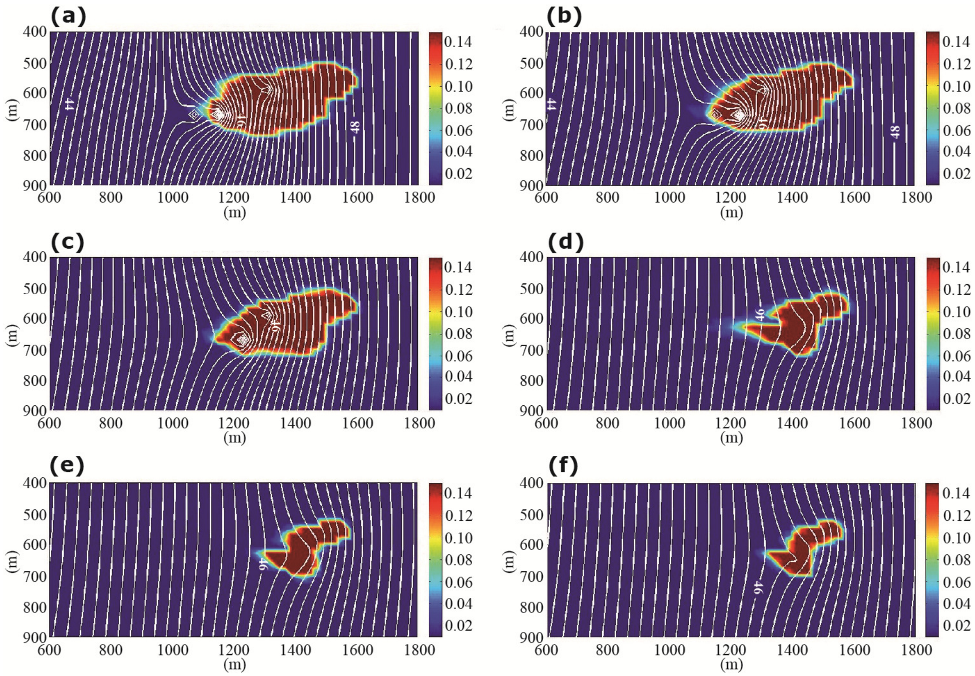


Figure 3.20 P&T remediation strategy. Annual evolution of contaminant concentrations (in mg L^{-1}) from remediation year 1 to remediation year 6 (e-f), and location of the extraction wells

Objective of the remediation strategy is to prevent the plumes from further spreading. The vertical lines presented in Figure 3.21 indicate the location at which the output concentrations of the simulation are recorded. At these locations, contaminant concentrations should not be higher than 0.150 mg L^{-1}.

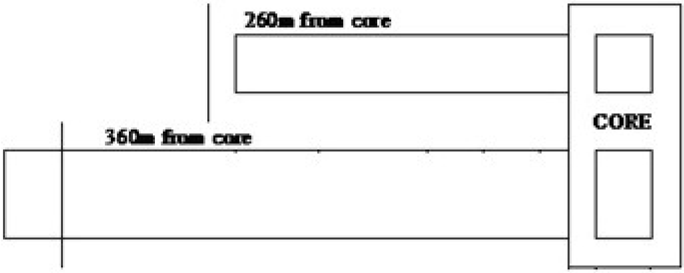


Figure 3.21 Location where output concentration are recorded

3.1.5.3 Results of the ACER and ICER: change in hydraulic gradient

For each remediation strategy, the evolution of maximum concentration at the southern plume, is presented in Figure 3.22. For the objective set, both the phytoremediation and P&T strategies succeed to capture the contamination. If no action is taken, the contamination first increases because contamination is still moving from the source zone towards the border of the side. Once the P&T starts operating at the source zone, concentrations decrease.

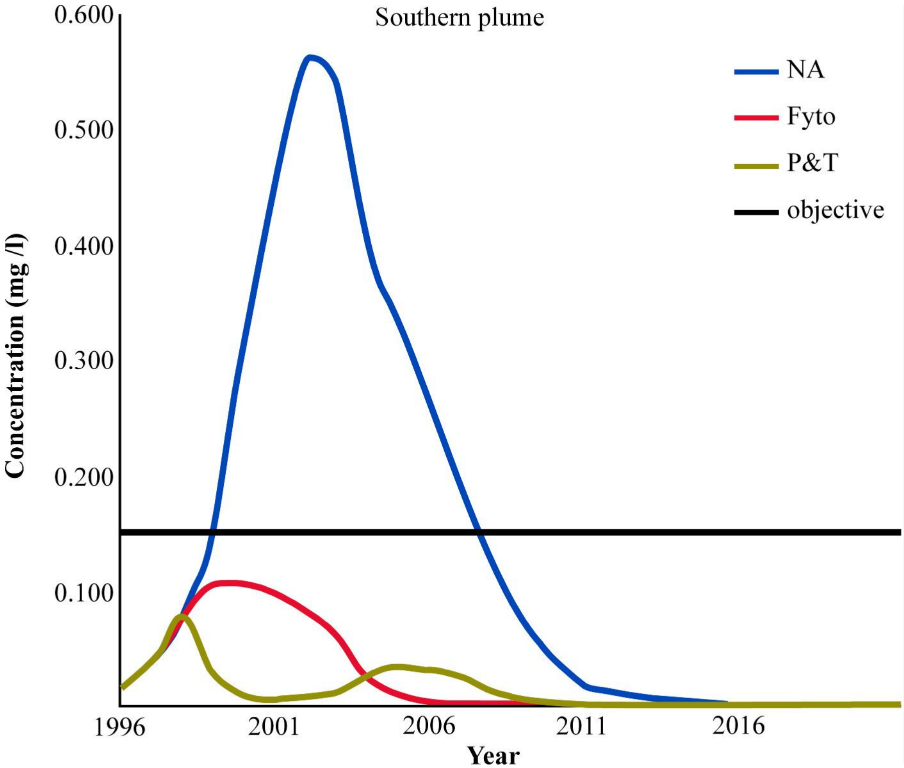


Figure 3.22 Evolution of maximum concentrations at the end of the southern plume

Total costs are calculated for a period of 20 years. A discount rate of 5% is applied. Assuming a gradient of 0.004, the contaminated area has become smaller: from 10 000 m² (gradient = 0.008) to about 5000 m². Therefore, it is assumed that less monitoring wells should be put in place and that less samples should be taken. Total cost equals €27 305 which is half of the amount presented in Table 3.4. Regarding phytoremediation, it is assumed that the same area of poplar trees should be planted. Only the monitoring costs decrease. The extraction wells for the P&T remediation strategy operate during a period of three years. The rental cost of the treatment unit decreases as less groundwater is extracted. Total costs for the P&T remediation strategy equal €84 239.

The ACER is calculated for different periods. After twenty years, the three remediation strategies reach the objective. However, from Figure 3.20 it is clear that the natural attenuation strategy is less effective. Both the P&T and phytoremediation strategy prevent the plumes from further spreading. During the first five years, P&T is more effective than phytoremediation. However, after 10 years the total amount of contaminant mass removed is the same for both P&T and phytoremediation, as a rebound effect occurs when the P&T strategy is applied.

Table 3.9 gives an overview of the different ACERs. If the remediation strategies would be evaluated after 5 years, the P&T strategy leads to the lowest cost per effect. Phytoremediation is the preferred remediation option when the effect is determined after 10 years. When the effect is determined after 20 years, the natural attenuation strategy results in the lowest cost per effect.

Table 3.9 Gradient 0.004: determination of the ACER

Remediation strategy	NA	phyto	P&T
Total discounted cost (€)	27 305.32	69 315.61	84 238.63
Mass removed after 5 years (kg)[a]	-1 476.00	51.20	216.00
Mass removed after 10 years (kg)[a]	-148.00	252.80	212.00
Mass removed after 20 years (kg)	256.00	256.00	256.00
ACER - 5 years	-18.50	1 353.82	389.99
ACER - 10 years	-184.50	274.19	397.35
ACER - 20 years	106.66	270.76	329.06

[a]A negative sign indicates an increase in concentration

The ICER is determined for an evaluation of the remediation strategy after 5 and 10 years (See Table 3.10). In the first case, the natural attenuation strategy is less effective but also less costly. P&T is more effective but also more costly. This result is visualized in Figure 3.23.

Table 3.10 Gradient 0.004: determination of the ICER

Remediation period	After 5 years		After 10 years	
Remediation strategy	NA	P&T	NA	P&T
ΔE (kg)	-1 527.20	164.80	-400.80	-40.80
ΔC (€)	-42 010.29	14 923.02	-42 010.29	14 923.02
ΔC/ΔE (€ kg^{-1})	27.51	90.55	104.82	-365.76

If the value of removing one extra kilogram of contaminated mass is valued less than €27, one should adopt the natural attenuation strategy to remediate the site. Phytoremediation is the preferred strategy if the value for a kilogram contaminated mass removed is in a range from €27 kg^{-1} to €90 kg^{-1}. If the value is higher than €90 kg^{-1}, P&T is the preferred remediation strategy.

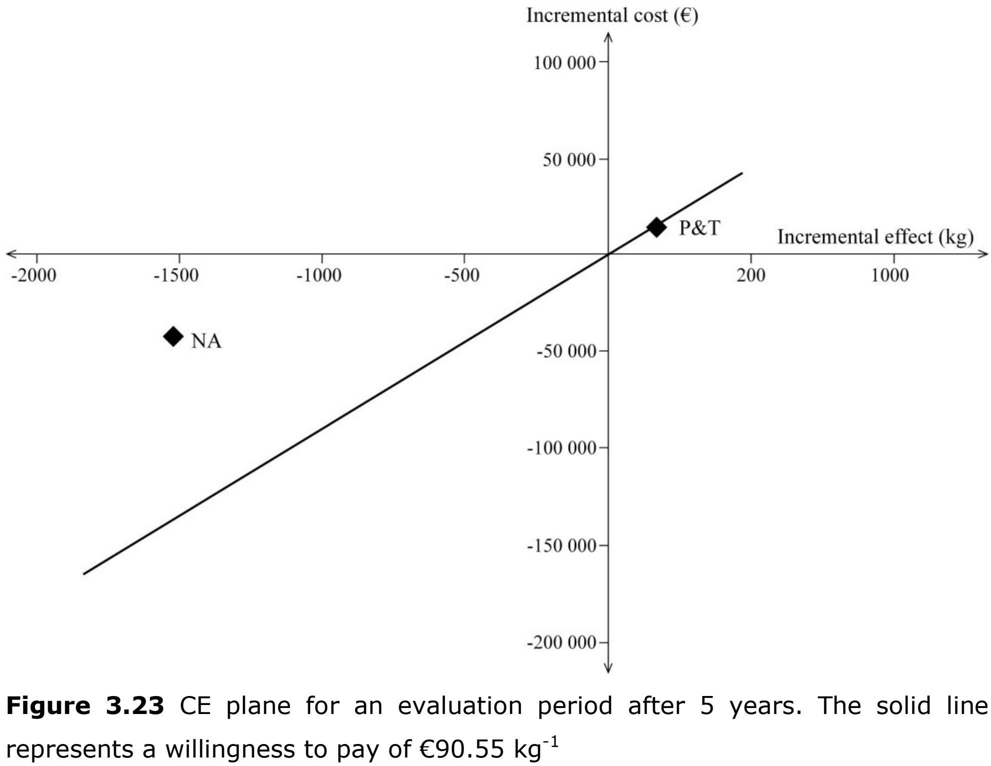


Figure 3.23 CE plane for an evaluation period after 5 years. The solid line represents a willingness to pay of €90.55 kg^{-1}

If the remediation strategies are evaluated after 10 years, the ICER of the P&T strategy moves to quadrant IV as this remediation option is less effective and more costly than phytoremediation (See Figure 3.24). The natural attenuation strategy should be selected if the value of removing an extra kilogram of contaminated mass is lower than €105 kg^{-1}.

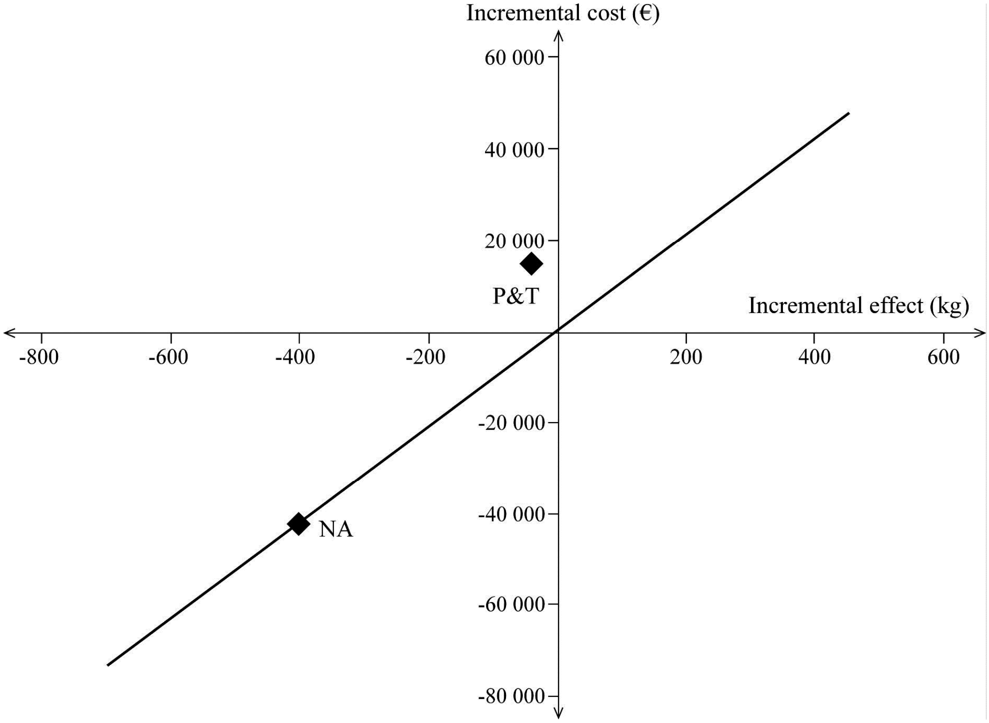


Figure 3.24 CE plane for an evaluation period of 10 years. The solid line represents a willingness to pay of €105 kg^{-1}

3.1.6 Pump & treat at the source zone

In practice, the P&T system at de source zone does not remove as much contamination as simulated. Whereas the simulation assumes a constant operation, the company indicates that the pump & treat system is often out of operation for repairs. The pump & treat system operates in a pulsed pumping mode, (alternating pumping and resting periods) which also results in a discontinuous removal of contaminants. The company regards the operation of the pump & treat system at the source zone as ineffective and expensive (after a 10 year operation period, this installation has cost more than €1000 000). The firm considers to stop its operation. The contamination at the source zone is situated 4 meters below a concrete surface and does not pose any risk to human health. Due to a groundwater velocity of 50 m year^{-1}, it is possible that the contamination will spread if the pump & treat system is shut down.

To determine the effect of shutting down the pump & treat system, the phytoremediation strategy as described in section 3.1.2.2 is simulated again, but without the pump & treat system at the source zone. Because the source weathering rate at the source zone is unknown, the simulation is run twice assuming a 12% (base case) and a 6% (worst case) source weathering rate. The evolution of contaminant concentration is compared with the results of the phytoremediation scenario presented in section 3.1.2.2. Figure 3.25 presents the evolution of contaminant concentrations at the source and phytoremediation zone for the different scenarios.

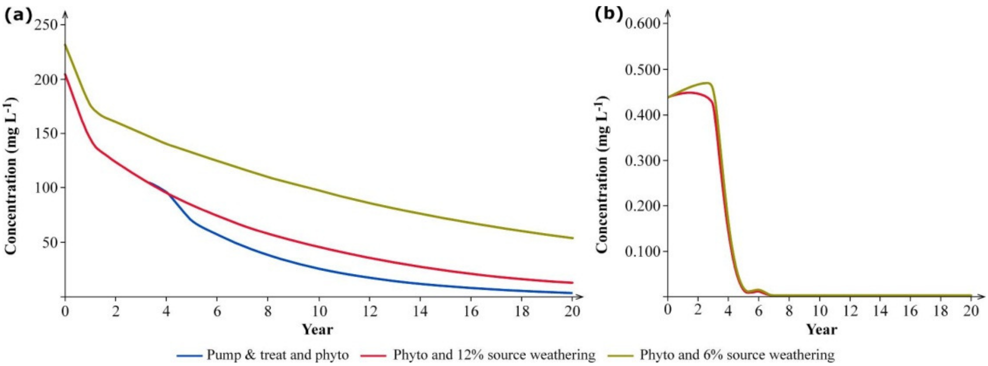


Figure 3.25 Evolution of the contaminant concentration (mg L^{-1}) at the source zone (a) and phytoremediation area (b)

Figure 3.25 (a) shows that without a pump & treat system at the source zone and a limited source weathering rate, the contamination at the source zone will reduce more slowly. Figure 3.25 (b) demonstrates that concentrations at the phytoremediation do not differ for the different scenarios. Only in the first years, when biodegradation is limited, the limited source weathering rate results in higher concentrations at the source zone.

3.1.7 Lessons learned

The aim of this study is to obtain a correct economic evaluation of the cost effectiveness of phytoremediation in comparison to other remediation technologies. A cost effectiveness analysis can be performed on an average or incremental basis, and different units of the effect can be considered. The use of these different calculation methods results in different conclusions. A definitive conclusion depends on how the firm or the authority prefers to express the effects, *e.g.* whether remediation time or whether quantity of mass removed is most important.

Different case studies will also lead to different results. The phytoremediation case study demonstrates that when the hydraulic gradient is limited to 0.004, the costs of the different remediation strategies is reduced because the surface of the contaminated site becomes smaller and less monitoring costs are required. Because concentrations are higher, more contaminant mass is removed which results in a smaller ICER. The extra cost of removing an additional unit of contaminant mass is smaller than when a gradient of 0.008 is applied.

Although the ICER better defines the trade-off between costs and effects compared to the ACER, it does not lead to a definite decision. In order to make a decision, the benefit of the remediation strategy should be determined. This study demonstrates how the cost of illness can be determined to calculate the remediation benefit, *i.e.* avoided health costs for the different scenarios. For the phytoremediation case study, it is shown that because of limited concentrations, natural attenuation is the preferred remediation strategy. However, the cost of illness is only related to the health risk associated with the contaminated site. Also the risk of further spreading of the contamination should be taken into account when evaluating the benefits of groundwater remediation.

This case study shows that a groundwater flow and transport model is useful to determine the effectiveness of different remediation strategies. Using the groundwater model, it is demonstrated that for this case study the pumping capacity of the trees is insufficient to capture groundwater flow and extract the contamination from the groundwater body. It shows the necessity to also introduce biodegradation rates. Moreover, the groundwater model makes clear that also the pump & treat system at the source zone does not operate sufficiently. The simulation of the pump & treat is based on reports of the firm. The total extraction rate at the source zone is 2.78 m^3h^{-1}, whereas it is determined that to capture the groundwater flow coming from the source zone, an extraction rate at the plume zone of $6m^3h^{-1}$ is required.

The groundwater flow model used is deterministic. This model does not address uncertainties regarding the effectiveness. In order to quantify the uncertainties associated with subsurface heterogeneity and to address the dynamics of flow and transport, stochastic groundwater modeling should be applied. Also note that the model results are produced for each discrete year. Regarding the phytoremediation case study, alternatives P&T and phytoremediation both reach the 150 mg L^{-1} objective after one year but the concentrations achieved are much lower when P&T is selected than when phytoremediation is adopted. Nevertheless, the results are robust, the phytoremediation area prevents the contaminant plumes from further spreading, also when the pump & treat system at the source zone is removed.

3.2 The cost effectiveness of bioremediation

3.2.1 Contaminant site description

As the result of former production operations and old leakages from storage tanks and pipelines, the aquifer at an industrial site in Belgium is contaminated with 1,2-DCA, an intermediate for industrial polyvinyl chloride production. The contamination covers an area of 99 125 m² and has spread more than 20 m in depth. The aquifer is composed of relatively homogeneous medium grained sands and the water table is situated 7 m below subsurface. The naturally occurring horizontal groundwater velocity is limited to about 1.7 m $year^{-1}$. This section will study the cost effectiveness of two remediation strategies: (i) pump & treat and (ii) bioremediation. Annex I provides a general description of these remediation strategies.

3.2.2 Simulation of remediation alternatives

To simulate the different remediation alternatives, a hydrogeological model is developed. Table 3.11 gives an overview of the hydrogeological input parameters.

Table 3.11 Hydrogeological parameters

Hydrogeological parameter	Value
Average horizontal conductivity (m d^{-1})	45
Average vertical conductivity (m d^{-1})	0.09
Longitudinal dispersivity (m)	0.08
Horizontal transverse dispersivity (m)	0.008
Vertical transverse dispersivity (m)	0.008
Retardation factor (dimensionless)	2.5
Porosity (dimensionless)	0.37
Decay (d^{-1})	0

The contamination and remediation at this site is divided in 2 areas: a southern plume and the factory area. Focus of this study is the remediation at the factory area. The bioremediation strategy involves a pump & treat system at the source zone of the contamination and 5 bioremediation cells will be installed at the east side of the area. The pump & treat system and the bioremediation cells will be operating simultaneously until concentrations at the source zone of contamination reach a level at which further operation of the P&T system is not cost effective anymore.

Pump & treat. At the source zone of the contamination, four extraction wells are put in place, indicated by the green dots on Figure 3.26. Groundwater is extracted at a total rate of 8 m^3 h^{-1}. The extracted groundwater is cleaned using granular activated carbon.

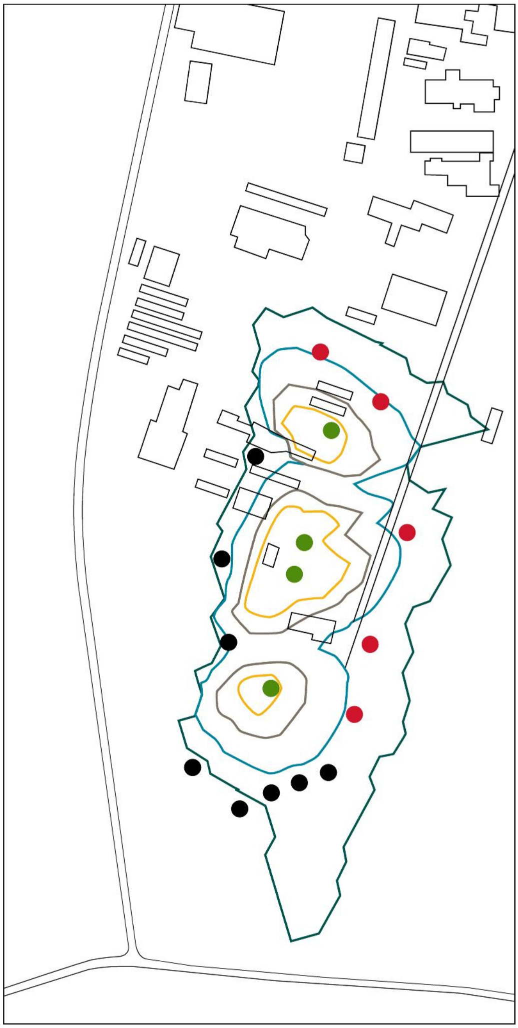


Figure 3.26 Location of the extraction wells for the P&T system (green dots), bioremediation cells (red dots), and recharge wells (black dots)

Bioremediation. The red dots on Figure 3.26 represent the five bioremediation cells. The bioremediation cells are situated at the eastern border of the site, preventing the contamination from further spreading. Each bioremediation cell consists of 4 extraction wells, 4 pumps, 2 recharge wells and a buffer tank. The configuration of the wells is shown in Figure 3.27. In a first phase, groundwater is extracted from the 3 extraction wells (EW1-3 in Figure 3.27), nutrients are added to the extracted groundwater and then the groundwater is recharged in two recharge wells (RW1-2 in Figure 3.27). Once the nutrients are spread between the recharge wells and the extraction wells 1-3, enriched and decontaminated groundwater is extracted from the central extraction well (CEW in Figure 3.27). The more groundwater enriched with nutrients reaches the central extraction well, the higher the probability that the contamination is degraded by the bacteria. The decontaminated groundwater extracted from the central well of each of the five bioremediation cells is injected into eight recharge wells. These wells are represented by black dots on Figure 3.26. Four of these wells form a hydraulic barrier at the southern part of the contamination. The four other recharge wells are located at the western border, creating a 'flush system'. To simulate the bioremediation strategy, the groundwater circulation from the central well to the eight recharge wells is modeled. From each central well, groundwater is extracted at a rate of 12.5 $m^3\ h^{-1}$. In total, 62.5 $m^3\ h^{-1}$ is extracted of which 50% is recharged in the hydraulic barrier and 50% is recharged at the western border of the site. Biodegradation is modeled by multiplying the contaminant concentrations in the groundwater extracted at the central well with a factor 0.4 which represent a biodegradation rate of 60%. This rate is the outcome of a model in which biological processes are simulated.

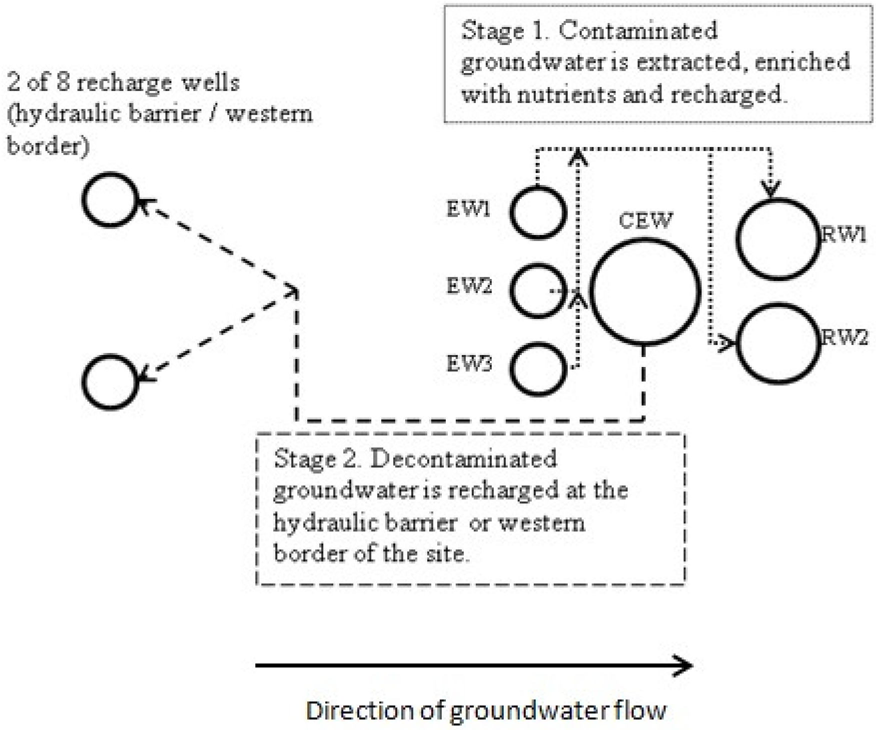


Figure 3.27 Bioremediation system: groundwater extraction and recharge. CEW = central extraction well, EW = extraction well, RW = recharge well

To decide how long the P&T system at the source zone should operate, different scenarios are developed. The bioremediation strategy is simulated for 20 years, the operation period of the P&T system is varied by 0, 2, 5, 10, and 20 years. For each of these scenarios, the costs and effects are determined.

The cost effectiveness of the bioremediation scenario is compared with the cost effectiveness of an alternative remediation strategy. This remediation strategy involves the same P&T system at the source zone combined with a P&T system at the north-eastern and eastern border of the site. Hence, all the extracted groundwater is treated above ground. Six extraction wells, presented by the red dots on Figure 3.28, are installed pair wise perpendicular to the groundwater flow to contain the contamination within these borders. The quantity of groundwater extracted is the sum of the average net recharge and the quantity of groundwater naturally flowing through the plume. Total extraction rate at the borders equals 7 m³ h^{-1}. At the source zone, groundwater is extracted at 8 m³ h^{-1}.

Figure 3.28 Location of the extraction wells at the source zone and the borders of the site

The containment strategy is simulated for a period of 20 years. The operation period of the P&T system at the source zone is varied, resulting in different scenarios. For each of these scenarios, the costs and effects are determined.

3.2.3 Results of the ACERs and ICERs: a bioremediation case study

3.2.3.1 Costs and effect

Table 3.12 gives an overview of the costs associated with the different remediation strategies. The investment cost for the five bioremediation cells includes the installation of well screens, pumps, pipe-work and a tank to store the nutrients. Also the cost of monitoring wells are considered. During the first year of operation, more nutrients will be added to the groundwater and therefore, annual costs are higher for the first year. The annual cost also includes groundwater monitoring. For the P&T system at the source zone, the company invests in well screens, pumps and a treatment unit. Annual costs include the usage of granular activated carbon, groundwater extraction, and maintenance. Regarding the containment strategy, first 15 $m^3 h^{-1}$ is extracted at both the source zone and eastern border. Therefore a larger treatment unit is required. At the end of its lifetime the treatment unit can be replaced by a unit that treats 7$m^3 h^{-1}$ if the company would decide to stop the extraction of groundwater at the source zone.

Table 3.12 Cost overview (€) for the different remediation strategies

Remediation strategy 1: bioremediation		Remediation strategy 2: Containment	
Bioremediation cells (62.5 $m^3 h^{-1}$)		P&T source+eastern border(15 $m^3 h^{-1}$)	
Investment cost	365 992	Investment cost	272 299
Annual cost year 1	136 027	Annual cost	164 329
Annual cost year 2-20	96 481		
P&T source zone (8 $m^3 h^{-1}$)		P&T eastern border (7$m^3 h^{-1}$)	
Investment cost	229 350	Investment cost	73 272
Annual operational cost	37 606	Annual cost	46 118

It is assumed that when the company decides to stop the extraction at the source zone, the extraction rate will be lessened and hence annual costs will decrease. For each of the two remediation strategies, 5 scenarios are developed. In each scenario, a different operation period (0, 2, 5, 10 or 20 years) of the P&T system at the source zone is considered. Table 3.13 presents an overview of total discounted costs. A remediation period of 20 years and a discount rate of 5% are considered.

Table 3.13 Overview of total discounted costs and the effectiveness of different remediation scenarios

Years P&T at source zone	0	2	5	10	20
Cost (€1000)					
Bioremediation	1 844	2 143	2 236	2 364	2 542
Containment	864	1 136	1 428	1 829	2 458
Effectiveness (ton)					
Bioremediation	137	159	174	183	183
Containment	52	55	56	57	58

The effect of each scenario is determined in quantity of contaminated mass removed. The bioremediation strategy proves to be more effective than the containment strategy as more contaminant mass is removed during a remediation period of 20 years. Concerning the containment strategy, the difference in operation period of the P&T system at the source zone has no impact on the quantity of contaminant mass removed. For the bioremediation strategy, the operation of the P&T system at the source zone raises the mass removal efficiency. However, there is no difference in mass removal for a P&T operation period of 10, or 20 years. Keeping the P&T system operational for a period longer than 10 years does not lead to more contaminant mass removed.

3.2.3.2 Average and incremental cost effectiveness analysis

To compare the cost and effect of the different remediation strategies and the associated scenarios, an average and incremental cost effectiveness analysis are performed. Table 3.14 gives an overview of the results. When the ACERS (C/E) of the bioremediation strategy are compared to those of the containment strategy, it is clear that the decision maker should select bioremediation combined with 5 years P&T at the source zone as the preferred remediation scenario. Concerning the calculation of the ICERs (ΔC/ΔE) the scenario bioremediation combined with 5 years pump & treat at the source zone is considered as the reference remediation strategy. Figure 3.29 presents the results of the ICER graphically. The scenario 'Cont 20y P&T' is dominated by the scenario 'biorem 5y P&T' as the latter is less expensive and more effective. For the other scenarios, the calculation of the ICER does not lead to an immediate decision.

Table 3.14 Calculation of the ACER and ICER for the containment and bioremediation strategy

Scenario	C/E (€1000 ton^{-1})	ΔC (€1000)	ΔE (ton)	ΔC/ΔE (€1000 ton^{-1})
Cont 0y P&T	16.78	-1 372.12	-122.24	11.23
Cont 2y P&T	20.64	-1 100.51	-118.71	9.27
Cont 5y P&T	25.55	-808.52	-117.85	6.86
Cont 10y P&T	32.34	-407.51	-117.18	3.48
Cont 20y P&T	42.75	221.24	-116.25	-1.90
Biorem 0y P&T	13.45	-392.16	-36.62	10.71
Biorem 2y P&T	13.49	-92.89	-14.89	6.24
Biorem 5y P&T	**12.87**	-	-	/
Biorem 10y P&T	12.93	127.57	9.12	13.99
Biorem 20y P&T	13.86	305.84	9.63	31.77

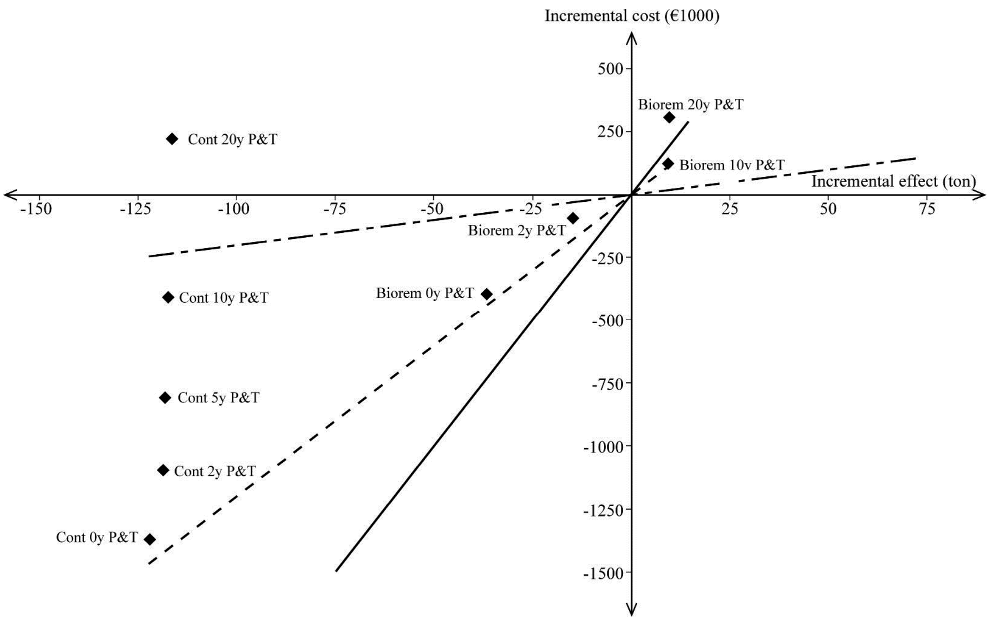


Figure 3.29 Cost effectiveness plane for λ = €2 000 ton^{-1} (dashed dotted line), λ = €12 000 ton^{-1} (dashed line), and λ = €20 000 ton^{-1} (solid line)

Also for this case study, different values of λ will lead to different results. If λ is smaller than €11 230 per ton, then the containment strategy with 0 years pump & treat should be selected. If λ is between €11 230 per ton and €13 990 per ton, then the reference scenario has the largest net benefit. For values of λ larger than €13 990 per ton, the largest net benefit is calculated for the bioremediation scenario with 10 years pump & treat. For each value of λ, the net benefit of the other scenarios is always smaller, these scenarios will never be selected. Table 3.15 gives an overview of the calculated net benefit.

Table 3.15 Net benefit (NB) in €1 000 for different λ

Strategy	λ = € 2 000 ton^{-1}	λ = € 12 000 ton^{-1}	λ = € 20 000 ton^{-1}
Cont 0y P&T	**1 127.65**	-94.72	-1 072.61
Biorem 5y P&T	-	**-**	-
Biorem 10y P&T	-109.33	-18.14	**54.82**

Figure 3.29 presents these results graphically. For instance, if λ is €2 000 per ton, the net benefit of the natural attenuation strategy with no pump & treat at the source zone is the largest. The ICER is situated most below the willingness to pay line. If λ is between €11 230 per ton and €13 990 per ton, then all the ICERs are situated above the willingness to pay line, the net benefit of the scenarios is negative. Hence, the reference remediation strategy should be selected. If λ is larger than €13 990 per ton, *e.g.* €20 000 per ton, then the net benefit of the bioremediation strategy with 10 years P&T is the largest.

3.2.4 Lessons learned

This case study confirms that economic results are site specific. For the bioremediation case study, total remediation costs are much higher compared to phytoremediation. Nevertheless, both the ACER and the ICER are much smaller because the contaminated volume removed is larger. Hence, substantial investment costs necessary to achieve clean-up is not an indication that the applied remediation strategy is not worth considering as a justifiable remediation option. Each remediation strategy available to remediate a specific site should be economically evaluated within the technological and hydrogeological boundary conditions, typical for that site.

Further, it is demonstrated that the cost effectiveness analyses cannot only be applied to select a remediation strategy, also the combination of different remediation strategies can be evaluated. Based on the results of both the ACERs and ICERs, the firm can decide whether it is cost effective to keep both remediation technologies operative. The option to abandon a remediation strategy when it proves to be ineffective is further studied in Chapter 4 and 5.

3.3 Conclusion and discussion

The case studies show how site characteristics, biological information, technological aspects and economics can be combined and analyzed to determine the cost effectiveness of the different remediation strategies considered. The site characteristics, and the technical properties of the different remediation strategies serve as input for the hydrogeological model. Based on the simulations, the effect of the different remediation strategies on groundwater flow and contaminant reduction is evaluated. The simulations give the firm a clearer insight into the problem and the results allow for a deliberate evaluation of the different effects. Further, it is shown how the results of the hydrogeological analysis are combined with cost data to evaluate the cost effectiveness of the different remediation strategy.

Concerning the cost effectiveness of phytoremediation and bioremediation, it is rather difficult to use these case studies to draw general conclusions because of the uniqueness of each contaminated site. However, this study provides an extended insight into the different calculations of cost effectiveness. Both case studies show that the calculation of the average cost effectiveness ratio (ACER) informs the decision maker on the overall efficiency of each remediation strategy, whereas the ICER takes into account the incremental costs and effects, leading to a more nuanced conclusion. Although a definitive decision based on the ICER can only be made when the willingness-to-pay for an incremental effect is known, the ICER better defines the trade-off between costs and effects compared to the ACER and a link to cost benefit analysis is provided.

The phytoremediation and bioremediation site conditions are completely different. As regards the site characteristics of the bioremediation case study, groundwater is polluted along a depth of 30 m and groundwater flow is limited. To clean up this site, extraction and recharge wells are configured to induce groundwater flow and to create a flushing system. Whereas for the phytoremediation case study, the contamination is more shallow, groundwater moves fast and should be captured in order to prevent the contamination from spreading further. Concerning the economic aspects, the bioremediation system requires a larger investment cost than when phytoremediation is applied, due to the use of extraction and recharge wells. Nevertheless, for the specific case studies, both phytoremediation and biostimulation can be considered as valuable alternatives among the remediation strategies available. Whether phytoremediation or bioremediation should be selected or not, depends on the benefit of the remediation.

In section 3.1.4, it is shown how the value of avoided health effects can be integrated in the analysis. If the remediation strategy’s objective is to prevent illness, phytoremediation is not the preferred remediation strategy in this case study. Concentrations are low, no health risk exists, hence natural attenuation should be selected. However, if the objective is to prevent the plume from further spreading, results can be different depending on the value of the damage incurred or not. The contaminated groundwater can reach surface water, a groundwater extraction well or cause damage to ecosystems.

Further, section 3.1.5 demonstrates that if concentrations are higher and if the area where concentrations are measured is located closer to the source zone, the results differ. Unlike natural attenuation, phytoremediation achieves to prevent the plume from further spreading. Moreover, when the remediation strategy at the source zone is out of operation (section 3.1.6), the phytoremediation strategy also prevents the contamination spreading further. In both cases, the reason that phytoremediation achieves the objective is due to biodegradation.

The knowledge on phytoremediation and bioremediation is limited and concentrated among a few scientists. In order to stimulate the adoption of phytoremediation and bioremediation, remediation experts or firms responsible for clean-up should learn about their merits. The procedures that exist nowadays to select an appropriate remediation strategy are overly simplified. Costs and effects are evaluated on a scale from 1 to 10, a true cost effectiveness analysis is never applied. Policy makers should develop a decision tool that allows for a correct comparison of cost and effects. The effectiveness of groundwater remediation strategies should be based on the results of a groundwater model, even though site characterization and the development of a groundwater model is time consuming and costly. Annex II of this chapter shows how valuable the use of a groundwater model can be. Further, the case studies show that for certain sites, phytoremediation can be considered as a feasible remediation strategy. Policy makers should stimulate remediation experts and firms responsible for clean-up to consider phytoremediation as a possible remediation alternative. Phytoremediation can be considered as the use of inexpensive solar powered pumps, well-fit in the framework of clean technologies.

3.4 Annex I Description of the different remediation technologies

3.4.1 Phytoremediation

Phytoremediation can be applied to both inorganic and organic contaminants. Inorganic contaminants can either be extracted from the soil by the plants and transferred to the shoots (phytoextraction) or immobilized by the plant’s roots (phytoimmobilization). For some inorganic and organic contaminants, it is possible that plants take up the contamination, transport it through the roots and transpire it into the atmosphere through the leafs (phytovolatilization). Within this dissertation, the phytoremediation approach considered is rhizodegradation. Rhizodegradation is the use of plants and plant-assisted microorganisms to degrade the contamination [92]. These microorganisms live in areas around the root of the plants. When phytoremediation is applied, the microbial community in the rhizosphere increases in number and uptake. Microorganisms can biodegrade a wide variety of organic contaminants [78], promote plant growth [12], and reduce evapotranspiration of volatile contaminants [20]. To successfully apply phytoremediation, root proximity, plant tolerance to contamination, and sufficient growth rates are conditions to be primarily fulfilled [11-18]. Despite these limitations, compared to traditional remediation technologies, phytoremediation is considered as a more environment-friendly technology that preserves soil fertility and structure. Due to high evapotranspiration rates, rapid growth, and phreatophytic root development, poplar trees are considered as ideal plants to remediate groundwater [18-20]. Poplars are inexpensive solar powered pumps that serve as hydraulic barriers [11, 15, 21, 22].

3.4.2 Natural attenuation

During natural attenuation, the pollutants are transformed to less harmful forms or immobilized. These processes are largely due biodegradation by microorganisms, and to some extent by sorption to the geologic medium. The natural attenuation process is contaminant specific and accepted as method for treating fuel components. Although remediation time might vary considerable with site conditions, natural attenuation is considered as a cost effective remediation strategy at polluted sites that do not require an aggressive remediation approach [24].

3.4.3 Pump & Treat

Pump & treat involves the extraction of groundwater and groundwater treatment on surface. The treated groundwater can be returned to the aquifer or discharged. The kind of treatment depends on the physical-chemical properties of the contaminants to be removed. A pump & treat system is designed to hydraulically contain and control the movement of contaminated groundwater and prevent continued expansion of the contamination zone. Or, pump & treat can reduce dissolved contaminant concentrations to comply with clean-up standards and thereby restore the aquifer. Pump & treat is most effective when the source of contamination is removed in order to prevent further contaminant introduction into the groundwater. Pump & treat effectiveness can be limited due to contaminant sorption onto the geologic media of the aquifer or by non-aqueous phase liquids (NAPLs) dissolution. NAPLs are lighter than water and tend to form a pool floating above the water table. Groundwater contamination results from the NAPL dissolving in the water and being transferred into the aquifer. Also problems of tailing and rebound are often observed when pump & treat is applied. Tailing refers to the slower rate at which contaminant concentrations decline after continued operation of a pump & treat system. Rebound is the increase in contaminant concentration that can occur after the pump & treat system is put out of operation [10].

3.4.4 Vertical engineered barrier

Vertical engineered barriers are most commonly slurry walls composed of native soils enriched with bentonite or another type of clay. This physical containment system is used to isolate contaminated soil or groundwater to prevent potential threats to human health and the environment outside the contained area. A groundwater extraction system is typically required to maintain inward and upward hydraulic gradients, so that the flux of water through the walls or floor is into the containment facility rather than out of it [7, 93]. Also a monitoring system is required to demonstrates that contaminants are not leaving the system at unacceptable rates. Although the concept of a containment system is relatively simple, implementation may be difficult. Successful construction of a wall that meets the design specification requires deployment of an experienced construction crew. Construction difficulties could create spots of higher hydraulic conductivity in some places in the wall, allowing for an outward flux of contaminants [93].

3.4.5 Permeable reactive barrier

The concept of a reactive barrier (PRB) involves the placement of reactive material in the subsurface where a plume of contaminated groundwater must move through it as it flows, typically under its natural gradient and treated water comes out the other side. Although the majority of installed PRBs use iron metal as the reactive media, also organic materials are being used to biologically remediate certain other contaminants. PRBs can be built in two basic configurations, the funnel-and-gate and the continuous PRB. The funnel-and-gate design uses impermeable walls as a funnel to direct the contaminant plume to a gate containing the reactive media. The continuous design completely transects the plume flow path with reactive media. The funnel-and- gate design has a greater impact on groundwater flow because of the funnels. For both designs, the permeability of the reactive zone should be equal to or greater than the permeability of the aquifer to avoid groundwater flowing around the reactive zone. To achieve successful clean-up, a thorough understanding of the nature of the contaminant, the reactive media and its chemical reaction, and the system hydrogeology is required [94, 95].

3.4.6 Biostimulation

Biostimulation is a remediation technology that involves the addition of electron acceptors, electron donors or nutrients to stimulate desired biodegradation reaction by encouraging the growth of more organisms, as well as by optimizing the environment in which the organisms must carry out the detoxification reactions [96, 97]. Whether microorganisms are successful in destroying man-made contaminants in the subsurface depends on the type of organisms, the type of contaminant, and the geological and chemical conditions at the contaminated site. Microbial transformation of organic contaminants occurs because the organisms can use the contaminants for their own growth and reproduction. Organic contaminants provide a source of carbon, and they provide electrons which the microorganisms can extract to obtain energy. The successful use of microorganisms in bioremediation is not simple. The technology requires knowledge not only of environmental engineering and hydrology, but also of the complex workings of microorganisms [97, 98]. Moreover, contaminant unavailability to the organisms, toxicity of the contaminant to the organisms, and partial degradation are key complicating factors that should be understood as well in order to achieve clean-up [97, 99].

3.5 Annex II The value of groundwater modeling

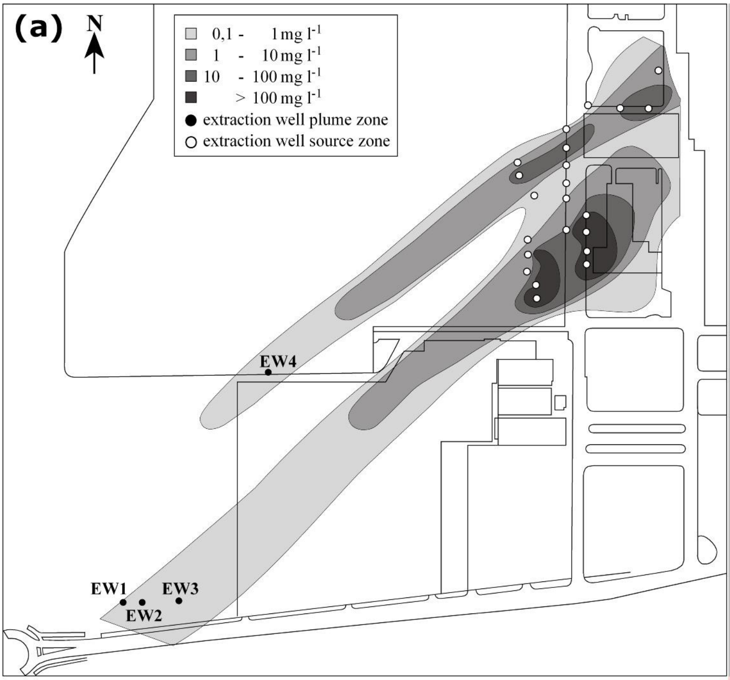
The simulation of groundwater flow can be based on a discrete set of initial conditions and parameter values *i.e.* deterministic simulations, or hydrogeological stochastic models can be applied.

The expense of data gathering [36] and the large computational effort required [37] are important reasons why stochastic modeling is not widely applied. Sometimes, not only the use of stochastic models is limited, also deterministic models are rarely applied to design remediation strategies. For instance in Flanders (Belgium), contractors mostly design by experience, consultants rely on rough-and-ready rules to determine well location and use an analytic approach to determine the number of wells and extraction rate [42].

To demonstrate how valuable the use of a groundwater model can be, it is necessary not only to look into the increased effectiveness of the designed remediation strategies, also an economic valuation of the groundwater model should be made. By describing a framework that presents the trade-off between the costs of developing a groundwater model and its benefits, the merits of a groundwater flow and transport model can be valued.

3.5.1 Problem description

The company located at the contaminated site in question, was advised to install three extraction wells at the end of the southern plume, perpendicular to the groundwater flow and one at the end of the northern plume as indicated in Figure 3.30(a). This remediation strategy however, was designed using rough - and ready rules. The groundwater model set up afterwards, showed that this design would fail and another P&T design was proposed (Figure 3.30(b)).



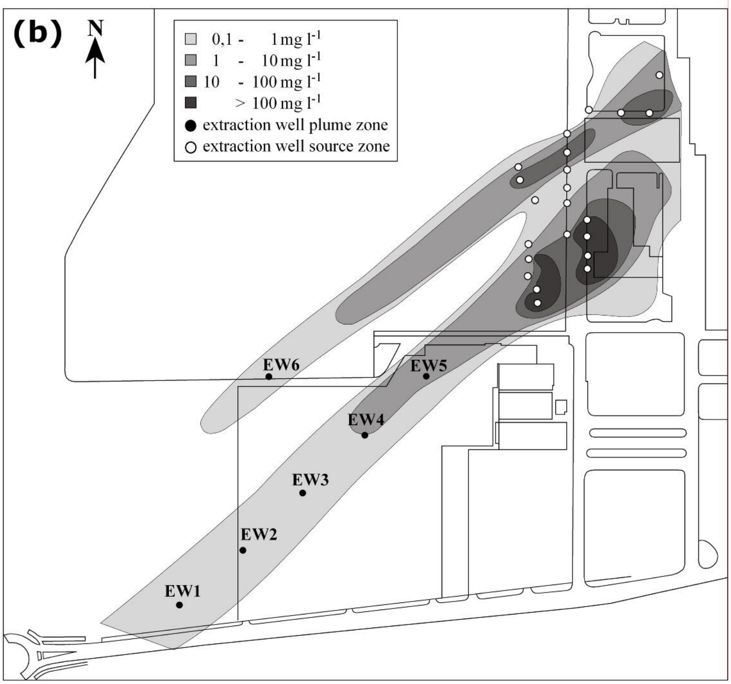


Figure 3.30 Design of the P&T system using rough –and ready rules (a) and using the hydrogeological groundwater model (b)

Figure 3.31 presents the difference in effectiveness between the two P&T designs. When the extraction wells are situated at the end of the southern plume, the contamination is pulled towards the wells. Concentrations increase until 2003 and then drop.

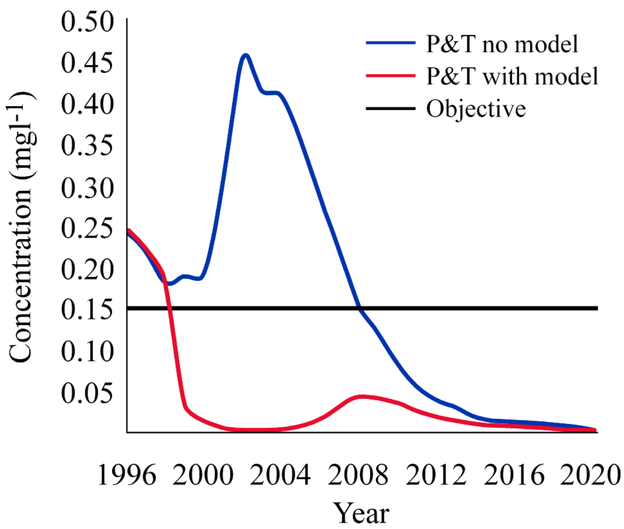


Figure 3.31 Evolution of contaminant concentrations for both P&T designs ($P\&T_{no\ model}$ and $P\&T_{with\ model}$).

The P&T system designed without using a groundwater model pulls the contamination towards the extraction wells. The remediation objective is only reached in 2009. The newly proposed P&T design reaches the objective already at the end of 1999 but should stay operative during 5 years.

The modelled concentrations presented in Table 3.16, show that when the rough –and ready design is applied, a remediation period of eleven years would be required to reach the remediation objective (0.150 mg L^{-1}). If the company uses the P&T system designed based on the hydrogeological model, the P&T system already reaches the remediation objective after one year. In order to avoid an unacceptable rebound effect, the P&T system has to stay operative during 5 years.

Table 3.16 Evolution of concentration at the end of the southern plume if the extraction wells are situated at the end of the southern plume or along the southern plume

Year	Concentration design using rough and ready rules (mg L^{-1})	Concentration design using the hydrogeological model (mg L^{-1})
1998	0.18	0.18
1999	0.19	0.04
2000	0.19	0.02
2001	0.31	0.01
2002	0.45	0.01
2003	0.41	0.01
2004	0.41	0.01
2005	0.35	0.01
2006	0.28	0.02
2007	0.22	0.03
2008	0.15	0.04
2009	0.12	0.04

3.5.2 Methodology

Within the economic framework, groundwater models are distinguished in terms of model complexity. It is assumed that the more input data and computer capacity is required, the more complex the model is, and hence, a higher model complexity will result in a higher model development cost. On the other hand, it is also assumed that due to an increase in information, a higher model complexity results in a lower probability of design failure and lower supplementary remediation costs due to design failure. Hence, remediation costs will decrease when model complexity increases. Moreover, due to an increased effectiveness, the remediation objective is more rapidly attained which can result in an earlier property sale. Also this benefit should be taken into account. The value of a groundwater model is equal to the net benefit (NB) of the model and is determined as follows [100]:

$$V(C_{model}) = NB(C_{model}) = B(C_{model}) - C(C_{model}) \quad (3.1)$$

With $V(C_{model})$ representing the value (or net benefit) of model complexity, $B(C_{model})$ the benefit of model complexity and $C(C_{model})$ the cost of model complexity. In the cost-benefit diagram presented in Figure 3.32, the horizontal axis represents the units of model complexity, starting at 0. This point could be seen as the use of rough-and-ready rules to define the P&T design. A movement to the right thus indicates an increase in model complexity. The benefit curve in Figure 3.32(A) equals the difference between the remediation cost when the quantity of complexity is equal to zero and the remediation cost at any given quantity of model complexity. The cost curve represents the difference between the cost of developing a model at any point of model complexity and the cost of using rough and ready rules to design a P&T system.

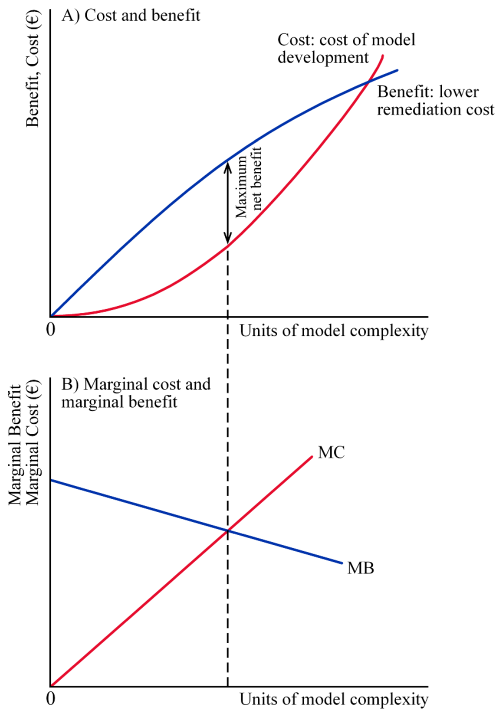


Figure 3.32 Cost benefit analysis of model complexity

When a groundwater model is developed, one should aim at maximizing the value of this model. To find the optimal level of complexity, the derivative of the value with respect to model complexity is set equal to zero:

$$\frac{dV(C_{model})}{dC_{model}} = \frac{dB(C_{model})}{d(C_{model})} - \frac{dC(C_{model})}{d(C_{model})} = MB_{C_{model}} - MC_{C_{model}} = 0 \tag{3.2}$$

The value of a groundwater model is maximized at the quantity of model complexity at which the difference between the benefit and the cost curve, the net benefit, is the largest. Model complexity should increase until the marginal benefit (MB) of a more complex model equals its marginal cost (MC). Figure 3.32(B) shows that the net benefit is maximized where the marginal benefit equals the marginal cost. In order to determine the net benefit (NB) of the groundwater model, total remediation costs for both the P&T designs (with and without model) and the total costs of model development should be calculated. The costs of a P&T system arise in different time periods. To make comparisons across time, future costs have to be discounted so that all costs are in a common metric: the present value. The present value is the current equivalent value of an amount that will be received in the future. In general, the present value (PV) of an amount Y received in n years with an interest rate i is calculated as follows [101]:

$$PV(Y) = \frac{Y}{(1+i)^n} \tag{3.3}$$

When costs occur in many periods, then the present value of the whole stream is calculated by adding the present values of the costs due in each period. If C_t denotes the costs occurred in period t for t = 0, 1, 2,..., n then the present value of the stream of costs, PV(C) equals [101]:

$$PV[C(C_{model})] = \frac{C_0}{(1+i)^0} + \frac{C_1}{(1+i)^1} + \cdots + \frac{C_{n-1}}{(1+i)^{n-1}} + \frac{C_n}{(1+i)^n} = \sum_{t=0}^{n} \frac{C_t}{(1+i)^n} \tag{3.4}$$

Similarly, if B_t denotes the benefit occurred in period t for t = 0, 1, 2, ...n then the present value of a stream of benefits, PV(B) is:

$$PV[B(C_{model})] = \sum_{t=0}^{n} \frac{B_t}{(1+i)^n} \tag{3.5}$$

Investing in a groundwater model is only worthwhile if the cost of establishing a groundwater model is smaller than its benefit. The net benefit (NB) determines whether a model should be introduced or not and is calculated as follows:

$$PV[NB(C_{model})] = PV[B(C_{model})] - PV[C(C_{model})] \tag{3.6}$$

A second decision rule that can be used to decide whether or not one should invest in a groundwater model is the Internal Rate of Return (IRR). To calculate the IRR, the use of a discount rate is not required. The IRR equals the discount rate (i) at which the NB is zero. One should only invest in a groundwater model if it earns a higher return than could be earned by investing the resources elsewhere. Note that the IRR is a percentage and does not take the scale of an investment into account. Two projects can have the same IRR but the NB of one project can be larger than the NB of the other project.

If total remediation costs of the P&T$_{\text{with model}}$ design are lower than those of the P&T $_{\text{no model}}$ design, then these savings can be perceived as the benefit of the groundwater model. Moreover, because the property can be sold sooner when the P&T $_{\text{with model}}$ design is adopted, the value of the property sale will be higher, and thus can be considered as a benefit as well.

3.5.3 Results

Table 3.17 presents the calculation of the total discounted costs for the implementation of the P&T $_{\text{no model}}$ design. Costs for coordination, wharf establishment, the installation of pumps and the (de)mobilization of the groundwater remediation unit occur only once and are called investment costs. Total annual costs equal €13 781 and include the use of granular activated carbon, pump rental cost and remediation unit rental costs. Total discounted costs are calculated applying Equation 4 and equal €137 235. A discount rate (i) of 5% is used.

The total discounted costs of the P&T$_{with\ model}$ design are calculated in Table 3.18. After each year, the pumps have to be changed and installed again. Each year, two pumps with an extraction rate smaller than 5 m³ and one pump with an extraction rate larger than 5m³ are operating. Using this design, more groundwater is extracted and therefore a larger groundwater remediation unit is required. Also the use of granular activated carbon increases compared to the previous design. Higher annual costs (€19 080) reflect these differences in design. Because less time is required to reach the remediation objective, the total discounted cost is lower (€97 780)

Table 3.17 Total discounted cost (€) of the P&T system designed without a model

Year	1999	2000	…	2008	2009
n	0	1	…	9	10
Coordination	5 402				
Wharf establishment	3 975				
Pipe works	1 425				
Mobilization remediation unit	1 364				
Installation pumps	4 875				
Granular Activated Carbon	4 351	4 351	…	4 351	4 351
Pump rental cost	3 674	3 674	…	3 674	3 674
Remediation unit rental cost	5 755	5 755	…	5 755	5 755
Total annual cost	30 823	13 781	…	13 781	13
Total annual discounted cost	30 823	13 125	…	8 883	8 460
Total discounted cost	**137 235**				

Table 3.18 Total discounted cost (€) of the P&T system designed with model

Year	1999	2000	…	2002	2003
n	0	1	…	3	4
Coordination	5 402				
Wharf establishment	3 975				
Pipe works	2 551				
Mobilization remediation unit	1 364				
Installation pumps	2 245	2 245		2 245	2 245
Cost for changing pumps	/	737	…	737	368
Granular activated carbon	6 527	6 527	…	6 527	6 527
Pump rental cost <5m³	1 837	1 837	…	1 837	1 837
Pump rental cost >5m³	1 473	1 473	…	1 473	-
Remediation unit rental cost	6 262	6 262	…	6 262	6 262
Total annual cost	31 636	19 080	…	19 080	17 239
Total annual discounted cost	31 636	18 172		16 482	14 182
Total discounted cost	**97 780**				

When the $P\&T_{no\ model}$ design would be applied, the property could only be sold in year 10 while in case of the $P\&T_{with\ model}$ design, the income from the property sale can be received in year 4. The property is about 30 000 m² large, and in 1999 the median price for industrial property equaled €12 m^{-2} [102]. Hence, the value of the property is determined at €360 000. However, the site needs to be remediated first. In case of the P&T no model design, the company would have to wait 10 years before the property can be sold. The present value of the property then equals €221 009. In case of a 5 year remediation period (P&T with model design), the present value of the property equals €296 173. This difference in revenue (€75 164) can also be perceived as the benefit of groundwater modeling.

The total benefit of the model equals €114 620 which is the sum of savings in the remediation costs (€39 455) and the increased value of the property sale (€75 164). In this case, investing in a model would only be worthwhile if the cost of establishing the model is smaller than €114 620.

Literature concerning the cost or price of model development was not found. An enquired expert stated that on average, €100 h^{-1} is asked, including administration, working hours and a consultancy fee. Developing a deterministic model would take about 150 hours. The cost of developing a model is an investment cost, due in year 0 and therefore discounting is not required. Using these numbers, the NB of model development is determined at €99 620. The calculation of the NB is shown in Table 3.19.

Table 3.19 Calculation of the Net Benefit (€)

PV(Costs)	**333 788**
Model investment cost	15 000
Remediation cost P&T $_{\text{with model}}$	97 780
Property sale P&T $_{\text{no model}}$	221 009
PV(Benefits)	**433 408**
Remediation costs P&T $_{\text{no model}}$	137 235
Property sale P&T $_{\text{with model}}$	296 173
PV(NB)	**99 620**

The costs of the P&T$_{\text{no model}}$ design is considered as a benefit of the model because this remediation technology is not executed anymore as consequence of the model development. Also the property sale in 2010 will not be carried out which is therefore a lost benefit and hence, a cost. The property sale in 2003 and the remediation costs of the P&T$_{\text{with model}}$ design are considered as the benefit and cost of the groundwater model respectively. The IRR or the discount rate at which the NB is zero, equals 40%. Also this decision rule indicates that investing in a groundwater model is worth the cost. The return earned on the investment is higher than the discount rate used in the calculation of the NB.

The net benefit is positive, and therefore should be visualized between the upper benefit line and lower cost line presented in Figure 3.32 (A). Because the net benefit represents only one point estimate, it is not possible to determine the maximum net benefit or to define the marginal cost or marginal benefit of developing this model. Nevertheless, one can conclude that developing a model can be worth its cost.

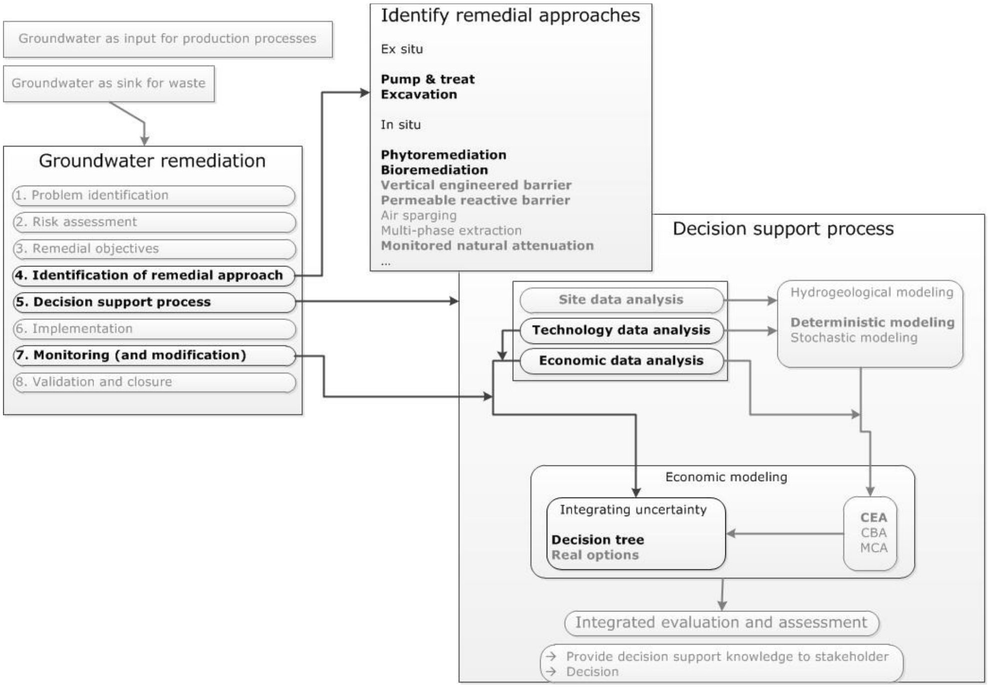
3.5.4 Lessons learned

Aim of this analysis is to provide a comprehensive framework that puts the discussion concerning the use of groundwater models in an economic perspective. A cost benefit analysis seems to be a suitable method to evaluate whether investing in a groundwater model is worthwhile. While before, the merits of using a groundwater model were mostly described in terms of technical effectiveness associated with the selected remediation technology. The quantification and monetisation of the impacts as consequence of model development has resulted in a defined economic value of the developed model. Within this economic framework, groundwater models are distinguished in terms of model complexity. This term is not further elaborated in this analysis but can be defined in hours required to develop the model and in terms of required computer capacity.

The economic theory is applied on one simple but clear case study: first, a P&T system was designed without using a model, afterwards a groundwater model was developed which led to a revision of the P&T design. Note that this case study concerns only a point estimate and although the model development resulted in a positive net benefit, it could not be verified whether the net benefit is maximized. Nevertheless, this analysis not only demonstrated that the correct use of groundwater flow and transport models can lead to better decision-making resulting in the application of a more effective and less expensive groundwater remediation strategy. It is also shown how valuable a model – and the advice of independent researchers – can be.

This *ex post* study, conducted after the groundwater model was established, also aims to provide information useful to predict costs and benefits in future *ex ante* analyses. It is precisely stated how costs and savings should be calculated and how a groundwater model can be valued. These calculations should aid parties responsible for groundwater clean-up in deciding about the use or non-use of groundwater models. Although this case study valued a deterministic model, the economic decision rules provided can also be applied to value stochastic models and optimization algorithms. The use of groundwater models to support decisions in groundwater remediation is only economically justifiable if the value of the extra information they provide exceeds the extra costs resulting from their application.

4 Dealing with removal efficiency uncertainty: a decision tree



4.1 Introduction

When making decisions concerning contaminated land management, the uncertainty inherent to the soil or groundwater problem, too, should be taken into account [4, 48]. The degree of knowledge of the site characteristics determines the success of cleaning up a contaminated site. Uncertainty related to both pollutant properties (*e.g.* contaminant strength and location) and hydrogeological parameters (*e.g.* horizontal conductivity) should be addressed in the decision analysis [48, 49]. Wang & McTernan [48] link the uncertainty associated with the underlying physical, chemical and hydrogeological analyses to the remediation costs and environmental risks. Stochastic modeling, risk assessment, simulation modeling and cost analyses are combined within a decision tree in order to identify an optimal remediation strategy for a given level of risk tolerance. This method results in a more robust decision making process, which enables the decision maker to be more aware of potential failure of the applied remediation technology. In this respect, Ma & Chang [49] developed a method to value the reduction in uncertainty by sampling the hydrogeological parameters in a groundwater remediation system.

The different types of economic evaluation tools mentioned above are effective decision support methods that are aimed at providing a concise representation of key decision making issues. However, these kind of economic evaluation tools do not consider the concept of decision reversibility.

Bage *et al.* [52, 53] developed a framework that considers the possibility of reevaluating the decision and selecting another remediation strategy. Focus is put on the selection of the optimal combination of site remediation technologies and the influence of uncertainty on the optimal strategy. In order to exemplify the proposed methodology, the remediation of a virtual contaminated site was simulated. This chapter further elaborates on the idea of redirecting the remediation as a means to address the removal efficiency uncertainty inherent to gentle remediation technologies, such as bioremediation.

Bioremediation, the use of microbial degradation processes to remediate soil and groundwater contamination, is perceived as a remediation strategy which has a considerable strength but also certain limitations. Factors such as strain selection, microbial ecology, the type of contaminant, environmental constraints, and also the design of engineered bioremediation systems may lead to the failure of bioremediation strategies [24, 25, 50]. The process of bioremediation involves complex and uncertain relationships among biomass, contaminants and nutrients. These uncertainties should be incorporated in the economic decision analysis. Wethasinghe *et al.* [103] developed an understanding of the impacts of uncertainty in biokinetic parameters on bioremediation, human health risk and economic decision analysis. Hu *et al.* [104] created a dynamic and multi-objective predictive control system that generates cost effective strategies for a bioremediation site involving uncertain data. However, in these studies too, the possibility to redirect the remediation strategy is not considered.

The uncertainty addressed in this chapter involves the unknown efficiency at which the groundwater contamination degrades when bioremediation is applied [105, 106]. Based on the bioremediation case study discussed in Chapter 3 (section 3.2), the value of a bioremediation project is compared with the value of applying Pump & Treat (P&T) of which the removal efficiency is assumed to be certain[2].

4.2 Simulation of groundwater flow and solute transport

The case study under consideration involves a fictitious contaminated site for which two remediation alternatives are available for clean-up. The decision maker can decide to adopt a pump & treat (P&T) system or to apply bioremediation in combination with P&T. The design of the bioremediation strategy is based on the design presented in section 3.2 but the considered groundwater body is fictive. Concerning the bioremediation process, different scenarios are simulated which reflect the uncertainty inherent to the application of bioremediation. In each scenario a different biodegradation rate is applied.

In order to evaluate the economic feasibility of bioremediation compared with P&T, the effectiveness of both remediation technologies should first be determined. A two-dimensional groundwater flow and transport model is developed using the code MOCDENS3D in which MODFLOW is integrated to calculate the groundwater flow [58, 59].

The finite difference grid consists of 5 rows, 100 columns and 10 layers. All cells of this grid have the same squared surface area with a side of 20 m and a height of 25 m. The difference in hydraulic heads results in a hydraulic gradient of 0.0025 that induces a limited groundwater flow of 0.02m d^{-1}. An annual net recharge of 0.28 m $year^{-1}$ is considered. Table 4.1 gives an overview of the hydrogeological input parameters applied.

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[2] Note that also P&T systems can fail to meet the specified objectives, due to an inappropriate design (see Annex Chapter 3) or unexpected rebound effects.

Table 4.1 Value of hydrogeological input parameters

Horizontal conductivity (m d-1)	8.00
Vertical conductivity (m d-1)	0.02
Longitudinal dispersivity (m)	0.30
Horizontal transverse dispersivity (m)	0.05
Vertical transverse dispersivity (m)	0.03
Retardation factor (dimensionless)	2
Porosity (dimensionless)	0.38
Gradient (dimensionless)	0.0025

In the centre of the defined groundwater body a volume of 475 000 m³ is contaminated with a concentration of 100 mg L^{-1}. A graphical representation of the groundwater body and the contamination (in grey) is shown in Figure 4.1.

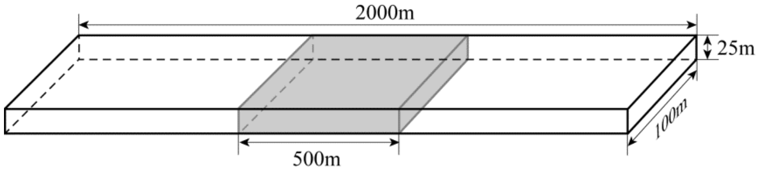


Figure 4.1 Graphical representation of the groundwater body and the contamination (in grey)

4.3 Simulation of remediation alternatives

Both P&T and bioremediation are simulated for a period of twenty years.

Strategy 1. With respect to the P&T remediation strategy, one extraction well with a screen length of 25 m is put in place in the centre of the contaminated area. This is presented by the dashed circle in Figure 4.2. The extracted groundwater is assumed to be cleaned above ground. In order to determine the groundwater extraction required to remediate the contaminated area within 20 years, the contaminated volume, net recharge (0.28 m y^{-1}), and the groundwater flow (0.02 m y^{-1}) are taken into account. To clean the contaminated volume (475 000 m³), groundwater should be extracted at a rate of about 5 m³ h^{-1} (taking into account a retardation of 2). To capture groundwater flow and net recharge, groundwater should be extracted at a rate of 4 m³ h^{-1}. This results in a total extraction rate of 9 m³ h^{-1}.

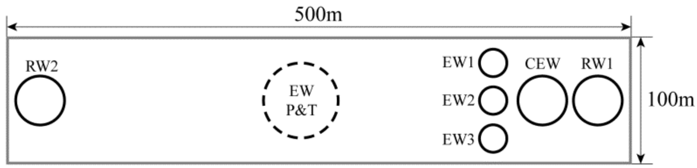


Figure 4.2 Top view of the contaminated area and the design of the two alternative remediation strategies. Design of the bioremediation strategy (recharge well and T-cell represented by the black circles) and the P&T strategy (one extraction well represented by the dashed circle). RW = Recharge Well, EW = Extraction well, CEW = Central Extraction Well.

Strategy 2. With respect to the bioremediation strategy, a combination of P&T and bioremediation is considered, similar to the configuration presented in Chapter 3. A configuration of one recharge well (RW2) and a T-bioremediation cell, as presented in Figure 3, is applied. The T- bioremediation cell consists of 3 extraction wells (EW1-3), one central extraction well (CEW) and one recharge well (RW1). The bioremediation T-cell operates as follows. In a first stage, nutrients are distributed around the central extraction well. Groundwater is extracted by the three extraction wells (EW1-3). The extracted groundwater is enriched with nutrients and injected in the recharge well (RW1). The nutrients then activate the bacteria present in the groundwater, which results in the degradation of the contamination. The central extraction well does not yet operate. In a second stage, the central extraction well extracts enriched and decontaminated groundwater, which is then injected in recharge well 2 (RW2) to induce an increased groundwater flow and flush the contaminated site.

The simulation of the bioremediation strategy only considers the hydrogeological aspects. Biological growth and degradation processes are not included in the simulation. Hence, in order to evaluate the effectiveness of the bioremediation strategy, only the second stage is simulated. This involves the extraction at the central extraction well and the injection of the enriched groundwater in recharge well 2. To simulate biodegradation, the concentration of the extracted groundwater is multiplied by a factor equal or smaller than 1. In order to know the impact of different factors, *i.e.* different biodegradation rates, the simulation is run six times. Each time a different factor is used: 0.00, 0.20, 0.40, 0.60, 0.80, 1.00. The factor of 0.00 equals a biodegradation rate of 100%, a factor of 1.00 equals a biodegradation rate of 0%.

The operation of the T-cell and the recharge well should lead to a clean-up of the contaminated area. Groundwater inflow resulting from net recharge and natural groundwater flow is captured by P&T. More groundwater (9 $m^3\ h^{-1}$) is extracted from the central extraction well (CEW) than is injected into recharge well 2. The recharge rate equals the rate that needs to be extracted to remediate the contaminated volume (475 000 m^3), *i.e.* 5 $m^3\ h^{-1}$ as explained previously. For the cost calculation, it is assumed that the extracted volume of groundwater which is not recharged, is cleaned above ground using granular activated carbon.

A decision tree structures the different actions to take. The decision maker can decide to adopt strategy 1 for twenty years or to adopt strategy 2 for twenty years. When an option to redirect the remediation strategy is included in the economic analysis, the decision maker has a third choice, *i.e.* to start up strategy 2 and after three years, if bioremediation proves to be inefficient, adopt strategy 1.

4.4 Results for the bioremediation case study

4.4.1 Costs and removal efficiency

In order to determine the removal efficiency of both strategies the annual average mass removed is calculated, taking into account the whole groundwater body. The average initial concentration equals 25 mg L^{-1}, which corresponds to a quantity of 95 ton contaminant mass initially present. Figure 4 shows the results of the simulations. With respect to bioremediation, a higher removal efficiency is registered when a higher biodegradation rate is applied in the simulation. The P&T simulation results in the highest removal efficiency.

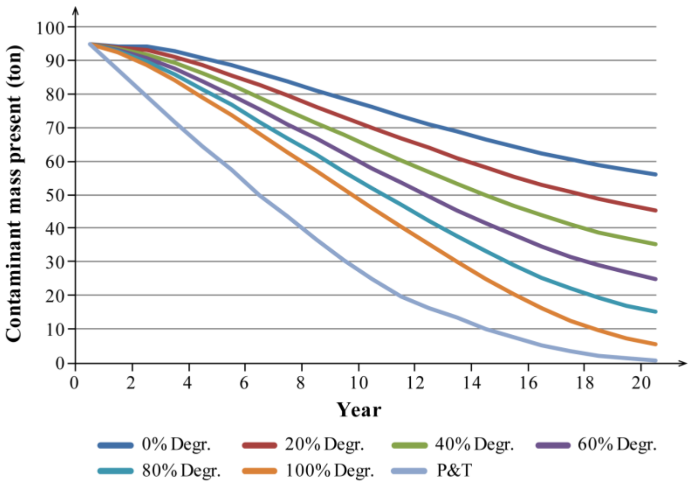


Figure 4.3 Simulation results for the P&T simulation and six bioremediation scenarios

The quantity of contaminant mass removed is considered as the benefit of the remediation strategy. Table 4.2 gives an overview of total quantity of mass removed.

Table 4.2 Overview of total quantity of mass removed

Simulation	Total quantity of mass removed (ton)
Degr. 0%	39.06
Degr. 20%	49.55
Degr. 40%	59.85
Degr. 60%	69.94
Degr. 80%	79.79
Degr. 100%	89.46
P&T	94.25

Table 4.3 gives an overview of the different costs resulting from the application of both technologies. These data are derived from the cost information provided

in Chapter 3 (section 3.2). With respect to the bioremediation strategy, the investment cost for groundwater extraction and recharge includes the installation of recharge and extraction screens, four pumps, a tank for substrate and the installation of monitoring screens. The annual operational cost includes site monitoring, utilities and supplies for the groundwater extraction and the use of substrate. In the first year, more substrate is used. Therefore, annual operational costs are higher in the first year. The treatment of 4 m³ h^{-1} above ground involves the investment cost for the groundwater treatment unit and an annual operational cost including the use of granular activated carbon and the utilities and supplies for the treatment unit. With respect to the P&T remediation strategy, the investment cost includes the installation of one screen, one pump and the groundwater treatment unit. The installation of monitoring screens is also included in the investment cost. For both remediation strategies, total costs are counted for a period of 20 years. It is assumed that well screens have a lifetime of 5 years. The pump and groundwater treatment unit should be replaced after 10 and 15 years respectively. A discount rate of 5% is used. In the Annex of this chapter a detailed overview of the different cost categories is presented.

Table 4.3 Overview of total discounted costs (€) for each remediation strategy

Remediation strategy 1: bioremediation		Remediation strategy 2: Pump & Treat	
Groundwater extraction (9 m³ h^{-1}) + recharge (5 m³ h^{-1})		Groundwater extraction + treatment (9 m³ h^{-1})	
Investment cost	95 810.00	Investment cost	142 084.22
Annual operational cost year 1	28 415.42	Annual operational cost	43 655.54
Annual operational cost year 2-20	20 739.86		
Groundwater treatment (4 m³ h^{-1})		Total discounted cost	709 212.33
Investment cost	59 374.86		
Annual operational cost	10 499.10		
Total discounted cost	591 437.47		

4.4.2 Calculation of the expected Net Present Value

The quantity of annual mass removed is considered as the benefit of each alternative. In order to attach a value to these quantities, the incremental cost effectiveness ratio (cf. chapter 3) for adopting the P&T alternative is calculated [107]. Table 4.4 shows the result. The difference in effectiveness is calculated by comparing the total quantity of contaminant mass removed under the P&T strategy with the average total quantity of contaminant mass removed under the different bioremediation contingencies, *i.e.* 65 ton. The value of €3 979 should be interpreted as the value at which one is indifferent between investing in P&T and bioremediation. To calculate the NPVs, the annual amount of mass removed is valued at €3 979. Under each bioremediation contingency, investment costs and annual operational costs are the same. Only the annual benefits differ due to the difference in removal efficiency.

Table 4.4 Calculation of the value for one ton of contaminant mass removed

	P&T
ΔC (€)	117 774.86
ΔE (ton)	29.60
ΔC/ΔE (€/ton)	3 979.20

Table 4.5 gives an overview of the NPVs of bioremediation under each contingency and the NPV of P&T. To calculate the expected NPV of bioremediation, the probability of occurrence is kept equal among the contingencies, *i.e.* 1/6 (0.17). The NPVs are calculated for a period of 20 years. A discount rate of 5% is applied. All NPVs are negative. It is assumed that only these two remediation technologies are applicable at the specified site and therefore, the strategy with the highest NPV should be selected. The expected NPV of bioremediation is larger than the NPV of P&T. Hence, when this evaluation tool is applied, bioremediation is the preferred remediation strategy. Note that for each contingency, the NPV of bioremediation is higher than the NPV of P&T.

Table 4.5 NPVs of the P&T and bioremediation strategy. The different contingencies associated with the bioremediation strategy are considered

Bioremediation contingency	NPV (€)
Degr. 0%	-486 815.10
Degr. 20%	-458 726.26
Degr. 40%	-431 136.14
Degr. 60%	-404 105.79
Degr. 80%	-377 725.41
Degr. 100%	-351 830.39
Strategy	NPV
bioremediation (expected value)	-418 389.85
P&T	-497 410.37

4.4.3 Decision tree

In order to include the possibility to redirect the remediation strategy, a decision tree is drawn. For this study, two stages are considered. If the decision maker starts with the application of bioremediation, after 3 years and for each bioremediation contingency, it is possible to adopt the P&T strategy for the remaining years. For the base case scenario, it is assumed that an investment cost of €15 000 is required to redirect the remediation strategy (from strategy 2 to strategy 1). This cost includes an upgrade of the treatment unit and the installation of the well screen. The cost of installing monitoring wells and a pump are excluded as these costs are made in the first year when strategy 2 is adopted. The annual cost of the remediation strategy also increases from €31 189 to €43 656. Figure 4.4 presents the decision tree and the different NPVs. In order to calculate the expected NPV of bioremediation, including the possibility to redirect the remediation strategy, the highest NPV under each contingency is considered *i.e.* NPV(1, P&T), NPV(2, BIOREM), NPV(3, BIOREM), NPV(4, BIOREM), NPV(5, BIOREM) and NPV (6, BIOREM).

The decision tree shows that if a biodegradation rate of 0% is registered after implementing the bioremediation strategy, it is preferred to redirect the remediation strategy and to adopt the P&T strategy. The expected value of bioremediation with option equals €-417 252, which exceeds the NPV of the P&T remediation alternative. The expected NPV is also larger than the expected value calculated in Table 4.5. Having the possibility to redirect the remediation strategy is worth €1 138. This is the difference between the expected value of bioremediation without option (see Table 4.5) and the expected NPV of bioremediation with option.

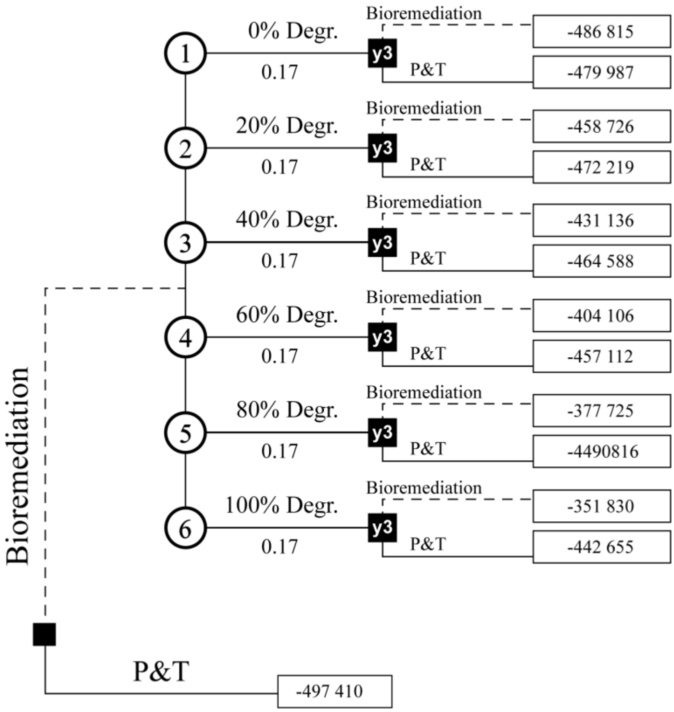


Figure 4.4 Decision tree assuming a value of contaminant mass removed of 3 979 € ton^{-1}. The remediation strategy can be redirected after year 3.

4.4.4 Sensitivity analysis

In the next section, the impact of changes in the specified assumptions on the expected NPV of bioremediation and P&T is examined. Figure 4.5A shows the impact of changing the value of the contaminant mass removed. When the value of the contaminant mass removed increases, the NPV of P&T and bioremediation increases as well. The option value, the area between the blue and red line, also increases. The intersection points are presented in more detail in Figure 4.5B. Bioremediation without option intersects P&T at a value of €6 841. When an option to redirect the remediation strategy exists, bioremediation intersects P&T at a value of €8 307. In case the value of removing one ton of contaminant mass were in the range between €6 841 and €8 307, the decision maker should only adopt bioremediation given that the option to redirect the remediation strategy exists.

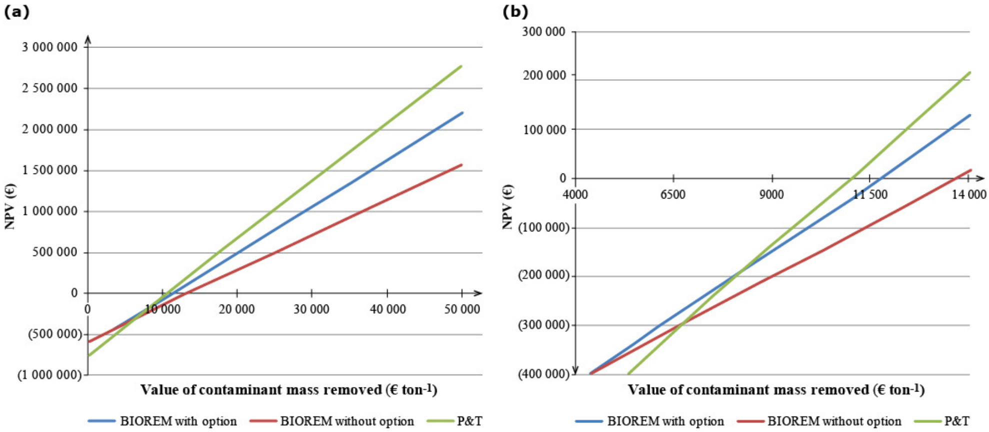


Figure 4.5 Effect of changes in the value of contaminant mass removed on the value of the different remediation strategies and the option value (area between blue and red line)

Table 4.6 gives a summary of the preferred remediation strategy given a certain range of values of contaminant mass removed (MAX NPV). As long as the value of one ton contaminant mass removed is smaller than €3 725, no option value exists.

Table 4.6 Overview of most preferable remediation strategy given the associated value ranges

Range (€ ton^{-1})	MAX NPV
0 – 3 725	Bioremediation without option = bioremediation with option
3 725 – 8 307	Bioremediation with option
8 307 - ∞	P&T

Figures 4.6A and 4.6B show the impact of variation in the probabilities attached to the contingencies associated with the bioremediation strategy. If the belief in a high removal efficiency is strong (a 70% probability is attached to the 100% degradation contingency), the value to have the option to redirect the remediation strategy is smaller than with a weak belief (a 70% probability is attached to the 0% degradation contingency). Table 4.7 presents an overview of the intersection points. When the decision maker has a strong belief in a high removal efficiency, a higher value of contaminant mass removed is required in order for the decision maker to select the P&T strategy. If the value of one ton contaminant mass removed is set at €3 979, the option value considering a strong and weak belief is €68 and €4 779 respectively.

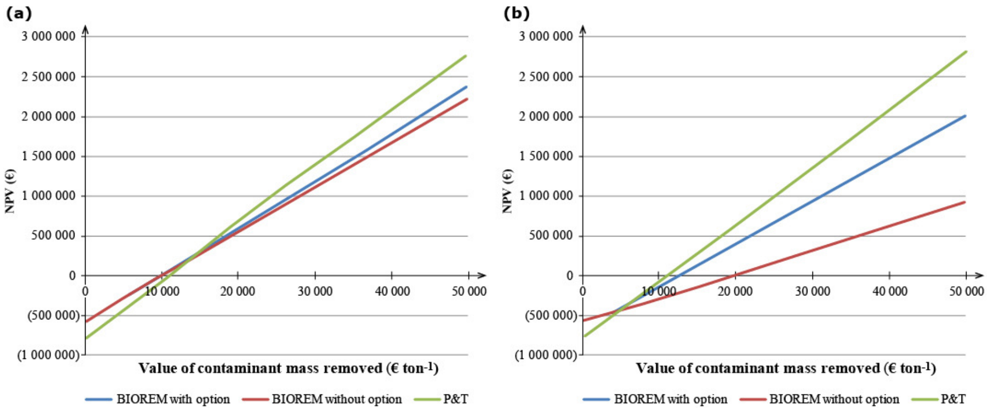


Figure 4.6 Impact of variations in a strong (a) and weak (b) belief in a high removal efficiency of bioremediation on the NPV of bioremediation, the NPV of P&T and the option value

Table 4.7 Intersection points for a strong and weak belief in a high removal efficiency

Strategy	Intersection value	Strategy	Intersection value
Strong belief		Weak belief	
Biorem with option	€14 137	Biorem with option	€5 540
Biorem without option	€12 879	Biorem without option	€4 631

When the point in time to redirect the remediation process is set later, *e.g.* after 10 years, the value to have the option to redirect the remediation strategy increases to €4 360. The decision tree is presented in Figure 4.7. The value of a ton contaminant mass removed equals €3 979. The probabilities of each bioremediation contingency are equally set at 0.17 (cf. Figure 4.4). The expected NPV of bioremediation including the possibility to redirect the remediation strategy equals -€414 030. NPV(1, P&T), NPV(2, P&T), NPV(3, BIOREM), NPV(4, BIOREM), NPV(5, BIOREM) and NPV (6, BIOREM) are considered.

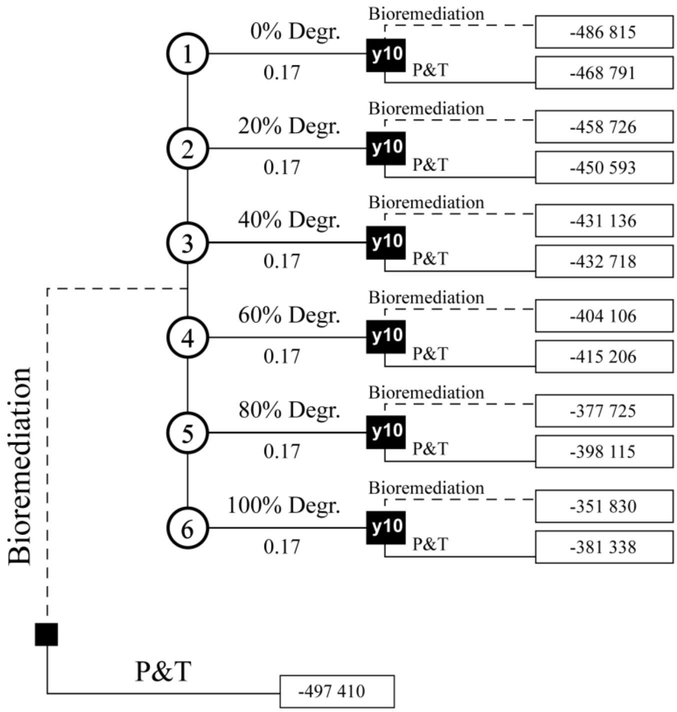


Figure 4.7 Calculation of NPV for the P&T and bioremediation strategy when the point in time to redirect the remediation strategy is set after 10 years

In the base case scenario, the investment cost required to redirect the remediation strategy is set at about €15 000. If the treatment system, the pump, and pipe-work cannot be upgraded or reused, then the installations present should be replaced, leading to an investment cost of €90 000. Figure 4.8 demonstrates that when a larger investment cost is required to redirect the remediation strategy, the option value decreases. Different values of contaminant mass removed are considered.

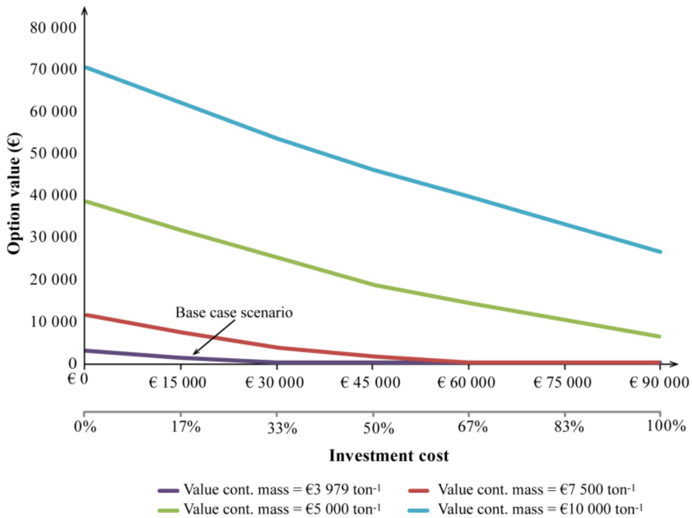


Figure 4.8 Effect of variation in the investment cost required to redirect the remediation strategy on the option value

4.4.5 Lesson learned

The analysis presented in this chapter indicates that there exist a value in the option to abandon bioremediation if it proves to be inefficient. Within the economic analysis presented, the decision maker has the possibility to redirect the remediation strategy, but only at one point in time. However, since multiple samples can be taken during a remediation processes, the parties responsible for remediation have the option to redirect the remediation strategy at any point in time. Therefore, more dynamic economic decision models that indicate the optimal timing of redirecting a remediation strategy should be developed (Chapter 5).

4.5 Conclusion and discussion

There exists a value in the option to abandon bioremediation if it proves to be inefficient. The NPV of the bioremediation strategy without option is smaller than then NPV of the bioremediation strategy with option. If the belief in an efficient remediation strategy is weak, the value of the option is higher than when the belief in an efficient remediation strategy is high. Further, it is demonstrated that when the value of the remediation increases, the option value increases as well. When the point in time to redirect the remediation process is set later, the value to have the option to redirect the remediation strategy is higher. Reason is the higher annual cost of the P&T strategy which occur later if the point in time to redirect the remediation strategy is set after 10 years instead of 3 years. If the investment cost required to redirect the remediation strategy increases, the option value decreases. In case of a high investment cost, it is less likely that the decision maker will redirect the remediation strategy.

With respect to the hydrogeological analysis, the complexity and uncertainty among biological processes associated with bioremediation are not modeled. However, the variation in biodegradation rates applied as input in the hydrogeological simulation can be considered as the output of models simulating these biological processes. The different biodegradation rates can be attached to different kinds of contamination (from inorganic to organic). Also note that the application of P&T is considered as a remediation strategy with a certain removal efficiency. However, also P&T systems can fail to meet the specified objectives, due to an inappropriate design (see Annex Chapter 3) or unexpected rebound effects [10]. Therefore, the authors argue that it would be useful to include uncertainty inherent to the application of P&T into the economic decision analysis.

For the hypothetical case considered, the probabilities used in order to draft the decision tree are selected to exemplify the outcome of a decision tree. A sensitivity analysis demonstrates the impact of changes in the assumptions on the outcome. For a real-life case study however, these probabilities should be based on historically observed frequencies or subjective assessments by experts based on information or theory [63].

Although the presented analysis is based on a fictitious case study, the designs of the different remediation alternatives and their associated costs are realistic. Accordingly, both the theoretical framework and the conclusions drawn from the economic analysis can be perceived as useful to future planned bioremediation projects.

Because there exist dynamic models that achieve better in defining the optimal timing to redirect the remediation strategy and the option value, the decision tree is only applied for one case study.

4.6 Annex I Cost overview of the remediation strategies

Tables 4.8 and 4.9 give a detailed overview of the costs considered for the calculation of the Net Present Values (NPV) for the bioremediation and P&T remediation strategy respectively. The treatment unit has a lifetime of 15 years. Because the period for which the NPVs are calculated is 20 years, the reinvestment cost for the treatment unit is considered to equal one third of the actual costs. For all cost categories, 10% unexpected costs are considered.

Table 4.8 Cost detail bioremediation strategy (€)

Strategy 1: Bioremediation	Units	Unit cost	Total
Investment cost			**155 185**
Monitoring			
Safety coordination	1.00	6000	6 000
Fixed cost	1.00	14000	14 000
Installation of monitoring screens	5.00	3500	17 500
Groundwater extraction and recharge (9m³ h^{-1})			
Installation of well screens	6.00	1800	10 800
Installation of pipe-work	420.00	40	16 800
Installation of pumps	4.00	4000	16 000
Installation of tank for nutrients	1.00	2500	2 500
Environmental and technical coordination	5.00	700	3 500
Groundwater treatment above ground (4m³ h^{-1})			
Installation of treatment unit	1.00	53 977	53 977
Unexpected cost (10%)			14 108
Annual cost year 1			**38 864**
Annual cost year 2-20			**31 189**
Groundwater extraction and recharge (9m³ h^{-1})			
Groundwater extraction and maintenance	1.00	8257	8 257
Groundwater monitoring	1.00	5500	5 500
Use of nitrates (year 1)	48.30	250	12 075
Use of nitrates (year 2-20)	20.39	250	5 097
Groundwater treatment above ground (4m³ h^{-1})			
Maintenance	1.00	4829	5 312
Mud disposal	1.00	1847	2 031
Granular activated carbon	1.00	1960	2 156
Unexpected cost year 1 (10%)			3 533

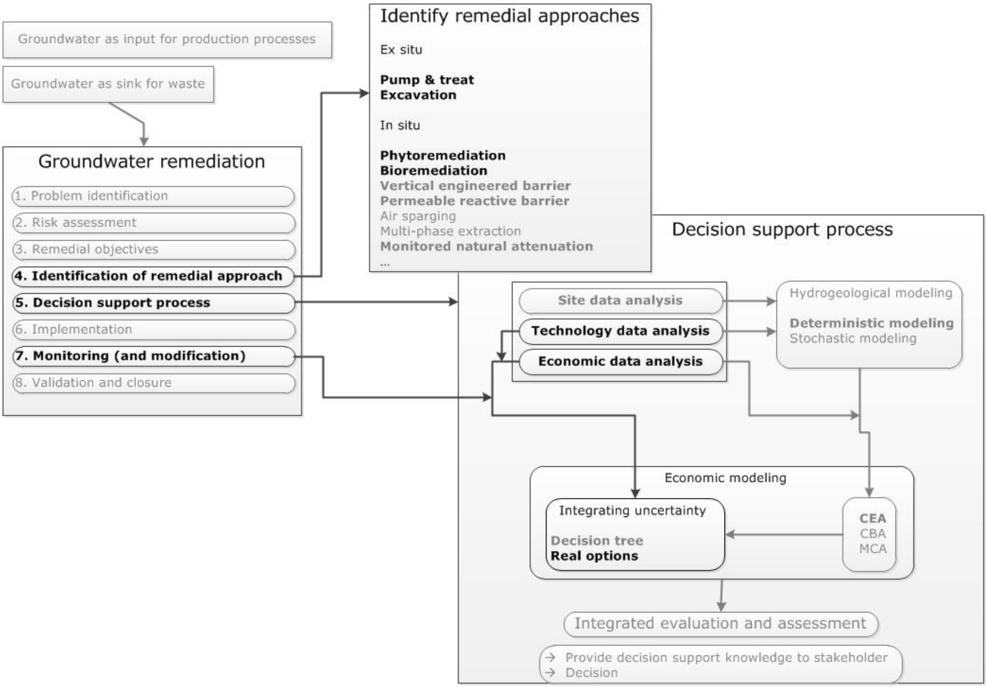
Unexpected cost year 2-20 (10%)			2 835
Reinvestment cost after 5 years			**7 577**
Recharge screens	2.00	1800	3 600
Pipe-work	50.00	50	2 500
Environmental and technical coordination	1.13	700	788
Unexpected cost (10%)			689
Reinvestment cost after 10 years			**33 963**
Recharge screens	2.00	1800	3 600
Extraction screens	4.00	1800	7 200
Pumps	4.00	4000	16 000
Pipe-work	50.00	50	2 500
Environmental and technical coordination	2.25	700	1 575
Unexpected cost (10%)			3 088
Reinvestment cost after 15 years			**28 010**
Recharge screens	2.00	1800	3 600
Pipe-work	50.00	50	2 500
Groundwater treatment unit	1.00	17 992	17 992
Environmental and technical coordination	1.96	700	1 371
Unexpected cost (10%)			2 546
Investment cost to redirect remediation strategy			**15 070**
Installation of well screen	1.00	1800	1 800
Installation of pipe-work	42.00	40	1 680
Installation of treatment unit	1.00	11 590	11 590

Table 4.9 Cost detail P&T strategy

Strategy 2: P&T	Units	Unit cost	Total
Investment cost			**142 084**
Monitoring			
Safety coordination	1.00	6000	6 000
Fixed cost	1.00	14000	14 000
Installation of monitoring screens	5.00	3500	17 500
Groundwater extraction (9m³ h^{-1})			
Installation of well screen	1.00	1800	1 800
Installation of pipe-work	420.00	40	16 800
Installation of pumps	1.00	4000	4 000
Installation of treatment unit	1.00	65 568	65 568
Environmental and technical Coordination	5.00	700	3 500
Unexpected cost (10%)			12 917

Annual cost			**43 656**
Groundwater extraction and recharge (9m³ h^{-1})			
Groundwater extraction and maintenance	1.00	10707	10 707
Groundwater monitoring	1.00	5500	5 500
Maintenance	1.00	13130	13 130
Mud disposal	1.00	5328	5 022
Granular activated carbon	1.00	1960	5 328
Unexpected cost (10%)			3 969
Reinvestment cost after 5 years			**3 616**
Pipe-work	50.00	50.00	2 500
Environmental and technical coordination	1.13	700	788
Unexpected cost (10%)			329
Reinvestment cost after 10 years			**10 863**
Extraction screens	1.00	1800	1 800
Pumps	1.00	4000	4 000
Pipe-work	50.00	50.00	2 500
Environmental and technical coordination	2.25	700	1 575
Unexpected cost (10%)			988
Reinvestment cost after 15 years			**28 235**
Pipe-work	50.00	50.00	2 500
Groundwater treatment unit	1.00	21 856	21 856
Environmental and technical coordination	1.88	700	1 312
Unexpected cost (10%)			2 567

5 Technical uncertainty and the option to abandon



5.1 Introduction

In Chapter 4, a decision tree is used to address the removal efficiency uncertainty inherent to a bioremediation strategy. It is demonstrated that it is valuable to have the option to redirect the remediation strategy if bioremediation fails to remove the contamination sufficiently. However, this decision tree does not define an optimal timing to abandon the remediation strategy. Dixit and Pindyck [54] developed the basic theory of irreversible investment under uncertainty, emphasizing the option-like characteristics of investment opportunities. The option theory takes into account the possibility to redirect a decision made and these authors illustrate that under uncertainty, the opportunity cost of not being flexible, rather than keeping the option open to rethink a project, is a significant component of the firm's investment decision.

5.2 Methodology

Dynamic programming is a general mathematical technology for the optimization of sequential decisions under uncertainty. A whole sequence of decisions is split into two components: the immediate decision and a valuation function that encapsulates the consequences of all subsequent decisions. The value function developed in Chapter 2 is applied for the same bioremediation case study presented in Chapter 4 and a new phytoremediation case study. Concerning the bioremediation case study, bioremediation is compared to a pump & treat system. As regards the phytoremediation case study, excavation is used as the alternative remediation strategy because excavation is actually applied at the site considered. From Chapter 2 we have that the value function is

$$V(k) = \begin{cases} V_1(k)\ if\ k > \ k^* + 1 \\ V_2(k)\ if\ k^* < k \leq k^* + 1 \\ \Pi_{strategy\ 2}\ if\ k \leq k^*, \end{cases}$$

with

$$\Pi_{\text{Strategy 2}} = \sum_{\text{t}=0}^{\infty} \frac{\text{annual net cash flow}}{(1+\text{r})^{\text{t}}} - \text{I}_{\text{Strategy 2}}, \tag{5.1}$$

$$V_1(k) = \frac{rBm_2^k + (1+r)a\lambda^k + (1+r)b\zeta(1-\lambda)^k}{r(\lambda^k + \zeta(1-\lambda)^k)}, \tag{5.2}$$

$$V_2(k) = \frac{Bm_2^{k+1}}{(r+1)(\lambda^k + \zeta(1-\lambda)^k)} + \frac{(r+\lambda)a\lambda^k + (r+1-\lambda)b\zeta(1-\lambda)^k}{r(\lambda^k + \zeta(1-\lambda)^k)} + \frac{\lambda(1-\lambda)(\lambda^{k-1} + \zeta(1-\lambda)^{k-1})}{\lambda^k + \zeta(1-\lambda)^k} \frac{\Pi_{\text{Strategy 2}}}{1+r}, \tag{5.3}$$

and

$$a = U_{Strategy1}^{H} - C_{Strategy\ 1}; b = U_{Strategy\ 1}^{L} - C_{Strategy\ 1}, \tag{5.4}$$

$$m_2 = \frac{(r+1)}{2} - \frac{\sqrt{(r+1)^2 - 4\lambda(1-\lambda)}}{2}, \tag{5.5}$$

$$B = \frac{\Pi_{\text{Strategy 2}} r\left[\lambda^{k^*}(\lambda + r) - \zeta(1-\lambda)^{k^*}(\lambda - r - 1)\right]}{\mathrm{r} m^{k^*+1}} - \frac{a\lambda^k\big(r + \lambda(1+r)\big) + b\zeta(1-\lambda)^k\big(r + (1-\lambda)(1+r)\big)}{\mathrm{r} m^{k+1}}. \tag{5.6}$$

p* is determined by

$$p(k^*) = \frac{X}{Y}, \tag{5.7}$$

where

$$X = \left(\mathrm{br} + \mathrm{b} - \Pi_{\text{Strategy 2}} r\right)[(\lambda-1)^2 + m_2(\lambda - r - 1) + r^2 - (\lambda - 2)r],$$

$$Y = m_2\big(a(r+1)(\lambda+r) - b(r+1)(-\lambda + r + 1) + (1-2\lambda)\Pi_{\text{Strategy 2}} r\big) +$$
$$(r+1)\big(-a(\lambda^2 + r^2 + \lambda r + r) + \mathrm{b}((\lambda-1)^2 + \mathrm{r}^2 - (\lambda-2)\mathrm{r}) + (2\lambda - 1)\Pi_{\text{Strategy 2}} r\big).$$

The threshold number of h-samples relative to l-samples is then given by

$$k^* = \frac{ln\left(\frac{p^*}{1-p^*}\right) + \ln(\zeta)}{ln\left(\frac{\lambda}{1-\lambda}\right)}. \tag{5.8}$$

5.3 The option to abandon: bioremediation

5.3.1 Site description

The case study under consideration involves the same fictitious contaminated site as presented in Chapter 4. The area is contaminated along a surface of 50 000m² and over a depth of 25m with a concentration of 100 mg per liter. This corresponds to 95 ton contaminated mass initially present (assuming a retardation factor of 2 and a porosity of 0.38). The decision maker can decide to adopt a biostimulation strategy or to apply pump & treat. In order to evaluate the effectiveness and hence, the revenues of the different remediation strategies, a groundwater flow and transport model is used to simulate both pump & treat and biostimulation. This model is developed using the code MOCDENS3D in which MODFLOW is integrated to calculate groundwater flow [58, 59]. In a first calculation, the value of the remediation strategies is determined without the option to abandon. Secondly, the real options theory is applied and the option to abandon the biostimulation strategy is integrated in the investment decision.

5.3.2 Results of the real options model: a bioremediation case study

With respect to the biostimulation strategy, the annual quantity of contaminant mass removed varies between 2.7 ton and 4 ton depending on the biodegradation rate, which varies between 0% and 100%. For the pump & treat strategy, the annual quantity of mass removed is determined at 4.7 ton. The valuation of the quantity of contaminated mass removed is based on a property

value of €23 m^{-2}. This value corresponds to the current average value of an industrial site in Flanders (Belgium). For a contamination rate of 0.019 ton m^{-2}, one ton of contaminated mass removed is valued at €12 105. It is also assumed that the decision maker has doubts about a highly efficient biostimulation project: p_0 equals 0.25. Groundwater quality is monitored four times a year, so the net cash flow of biostimulation and the discount rate are determined on a quarterly basis. The net cash flows for a high and low efficient biostimulation project are represented by a and b respectively.

If the decision maker does not consider the option to redirect the remediation strategy, then the value of the investment in biostimulation (biostim) equals

$$\mathrm{NPV}_{\text{biostimulation without option}} = \mathrm{p(k)}\sum_{\mathrm{t=0}}^{\infty}\frac{\mathrm{a}}{(1+\mathrm{r})^{\mathrm{t}}} + (1-\mathrm{p(k)}\sum_{\mathrm{t=0}}^{\infty}\frac{\mathrm{b}}{(1+\mathrm{r})^{\mathrm{t}}} - \mathrm{I}_{\text{biostim}}$$

$$= \frac{(1+\mathrm{r})\mathrm{a}\lambda^{\mathrm{k}}+(1+\mathrm{r})\mathrm{b}\zeta(1-\lambda)^{\mathrm{k}}}{\mathrm{r}(\lambda^{\mathrm{k}}+\zeta(1-\lambda)^{\mathrm{k}})} - \mathrm{I}_{\text{biostim}} = -€18\ 263,$$

with $a = €4\,207; b = €382;\ \lambda = 0.7;\ k = 0;\ r = 0.0123; I_{biostim} = €128\,667;\ \zeta = \frac{1-p_0}{p_0}$.

The NPV of the investment in pump & treat (P&T) equals

$$\mathrm{NPV}_{\mathrm{P\&}T} = \sum_{\mathrm{t=0}}^{\infty}\frac{\text{annual net cash flow}}{(1+\mathrm{r})^{\mathrm{t}}} - \mathrm{I}_{\text{pump \& treat}} = \text{annual net cash flow} * \frac{1+\mathrm{r}}{\mathrm{r}} - \mathrm{I}_{\text{pump \& treat}}$$

$$= €24\ 498,$$

with $annual\ net\ cash\ flow = €7\,555; I_{pump\,\&\,treat} = €142\ 084, r = 0.05.$

If the decision maker does not take into account the possibility to redirect the remediation strategy, the decision maker will decide to adopt the pump & treat system.

If the decision maker integrates the option to adopt pump & treat if biostimulation proves to be inefficient within the decision making process, then the value of the remediation project equals,

$$\frac{rBm_2^k + (1+r)a\lambda^k + (1+r)b\zeta(1-\lambda)^k}{r(\lambda^k + \zeta(1-\lambda)^k)} - I_{biostim} = €60\ 449$$

with $a = €4\,207; b = €382;\ \lambda = 0.7;\ k = 0;\ r = 0.0123; I_{biostim} = €128\,667;\ \zeta = \dfrac{1-p_0}{p_0};$

$B = 3\ 720\ 211, m_2 = 0.2913.$

Figure 5.1 shows the different results. When the option to redirect the remediation strategy is not taken into consideration, the decision maker will adopt the pump & treat strategy. If the option to abandon the biostimulation strategy is included, the decision maker will invest in the biostimulation strategy.

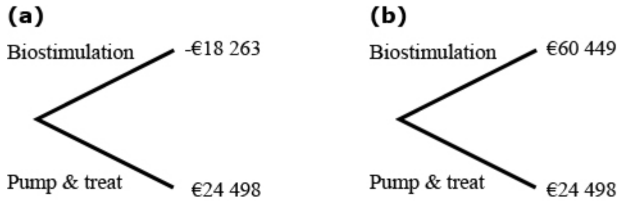


Figure 5.1 Decision tree without the option (a) and with the option (b) to redirect the remediation strategy

To determine the optimal timing at which the decision maker should adopt pump & treat if the biostimulation strategy proves to be inefficient, k* is calculated. As it is required to start the biostimulation strategy in order to evaluate its removal efficiency, it is assumed that the firm has already made the investment associated with the adoption of biostimulation. Hence, to determine k* the investment cost of the biostimulation strategy is considered as a sunk cost. In case the firm has to redirect the remediation strategy, it is assumed that €15 000 should be invested to adopt pump & treat.

K* is determined at -2.82. Figure 5.2 presents the value function as a function of k graphically. It is demonstrated that since k* = -2.82, it is optimal to redirect the remediation strategy and adopt the pump & treat strategy as soon as the number of bad samples is 3 higher than the number of high samples. The value functions V_1 and V_2 intersect at k*+ 1.

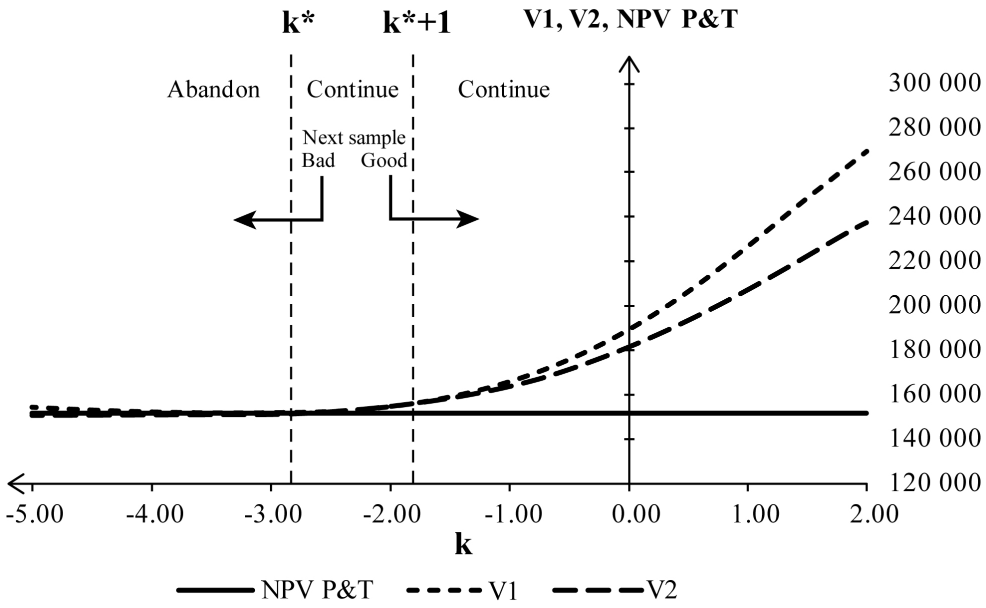


Figure 5.2 Value function (in €) as function of k

p(k*) equals 0.03 which means that as long as the belief in a highly efficient biostimulation strategy is higher than 3%, one should continue biostimulation. p* depends on λ. Figure 5.3 a and b demonstrate that the larger the probability that the signal is correct, the lower the critical belief in a good remediation project. However, still less bad samples in excess of good samples are required to redirect the remediation strategy. For the base case scenario, λ is set at 0.7 which reflects a rather correct signal. As a result, only a few bad samples in excess of good samples need to arrive to redirect the remediation strategy and the belief in a good biostimulation project drops from 0.25 to 0.02. At that state, the firm is convinced that the biostimulation project is inefficient. If the samples do not reflect the true state of the remediation strategy (λ=0.5), the signal does not provide any information. The value of the critical belief is then determined as follows,

$$\lim_{\lambda=0.5} p^* = \frac{\frac{r\Pi_{lock-in}}{1+r} - b}{a-b} = 0.38$$

Because p* is larger than p_0 (0.25) for λ = 0.5, the company will never adopt biostimulation if the samples taken do not provide any reliable information. If p_0 would be larger than p* (for λ = 0.5), then the decision maker will never redirect the remediation strategy. The relation between p*, p_0 en k* for λ = 0.5 can be described as follows,

$$If\ 0 < p_0 < p^*, then\ \lim_{\lambda=0.5} k^* = +\infty$$
$$If\ p^* < p_0 < 1, then\ \lim_{\lambda=0.5} k^* = -\infty$$
$$If\ p_0 = p^*, then\ \lim_{\lambda=0.5} k^* = 0$$

If the samples taken do not provide any information (λ = 0.5), the firm is indifferent between biostimulation and pump & treat if its initial belief in a good biostimulation project equals the value of the critical belief. If its initial belief in an efficient biostimulation project is smaller than the critical belief, the firm will never consider biostimulation, as its belief cannot be updated due to the bad quality of the samples. If its initial belief is larger than the value of the critical belief, the decision maker's belief will not be changed and he will continue with the biostimulation strategy.

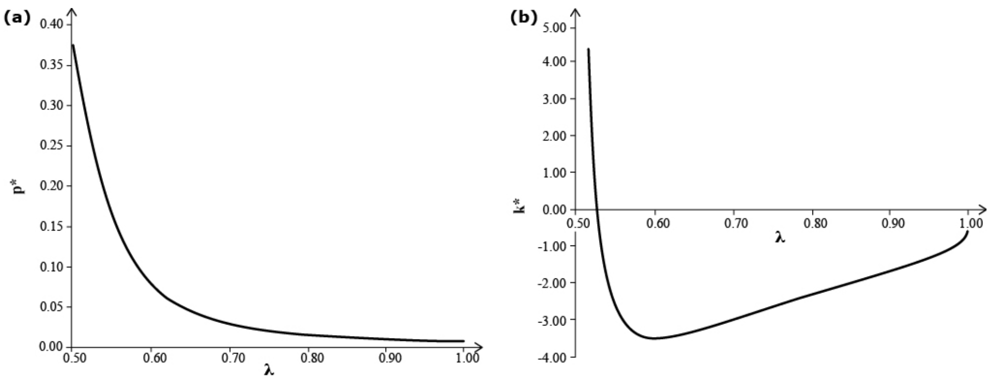


Figure 5.3 p* as a function of λ (a) and k* as a function of λ with p0<p* for λ=0.5 (b)

Figure 5.4 presents V_1 as a function of k, separated into the option value and the value of the biostimulation strategy. The more biostimulation proves to be inefficient, the more valuable it is to have the option to redirect the remediation strategy. The option value is very low when the number of good samples in excess of bad samples is higher than 6.

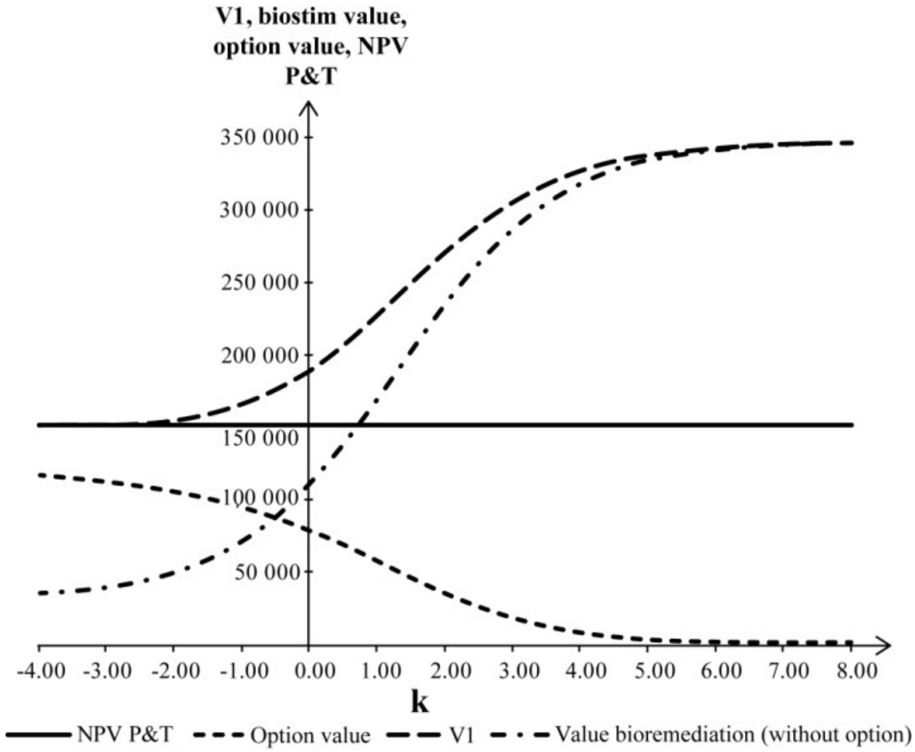


Figure 5.4 Presentation of V1, the *option value* and the biostimulation value as a function of k

5.3.3 Sensitivity analysis

For the base case scenario, the valuation of groundwater remediation is based on a property value of €23 m^{-2}. This value can vary geographically or can be different for different types of land use. Therefore, the effect of variations of this parameter value on k* is examined. Figure 5.5 makes clear that if the value is smaller than €18 m^{-2}, it will never be necessary to invest in pump & treat,

because then the bioremediation strategy dominates. If this value is higher than €43 m^{-2}, the decision maker will immediately adopt the pump & treat strategy.

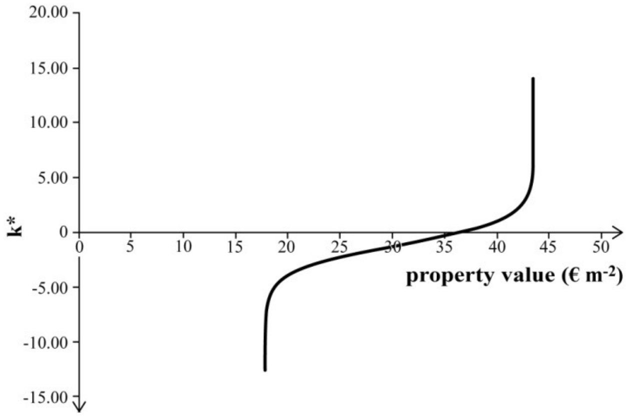


Figure 5.5 Effect of variations in the value of one ton contaminant mass removed on k*

Figure 5.6 shows that the effect of variation in the investment cost on k* is limited as long as the investment cost is smaller than €135 000. Once the investment cost would be larger than €135 000, the company will never adopt the pump & treat remediation strategy.

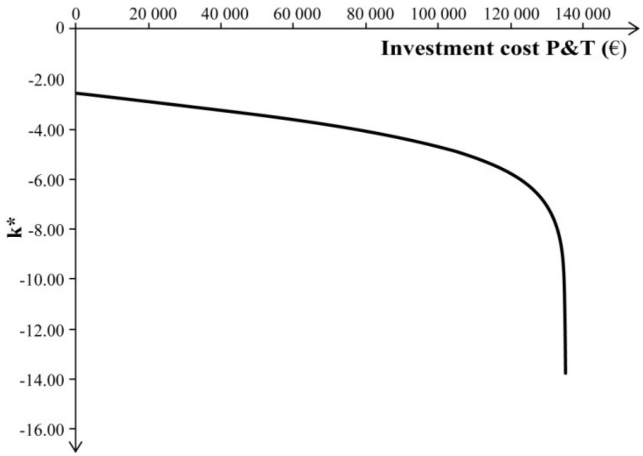


Figure 5.6 Effect of variations in the pump & treat investment cost on k*

For the base case scenario, the efficiency of the biostimulation project varies between 2.7 and 4.0 ton of contaminant mass annually removed. Figure 5.7 shows that if the upper limit would drop to less than 3.14 ton, the decision maker should immediately adopt pump & treat and never consider the biostimulation strategy.

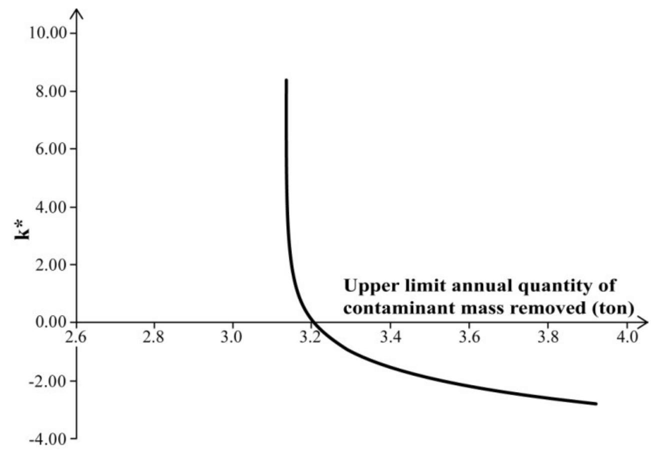


Figure 5.7 Effect of variations in the upper efficiency limit on k*

Concerning the initial investment decision, p_0 is set to 0.25 for the base case scenario. Figure 5.8 demonstrates how the initial investment decision varies for different values of p_0. If p_0 is larger than 0.39, the decision maker invests in the biostimulation strategy, even if the option to abandon does not exist. If the initial belief in an efficient biostimulation strategy varies between 0.05 and 0.39, the option to redirect the remediation strategy is required for the decision maker to invest in the biostimulation strategy.

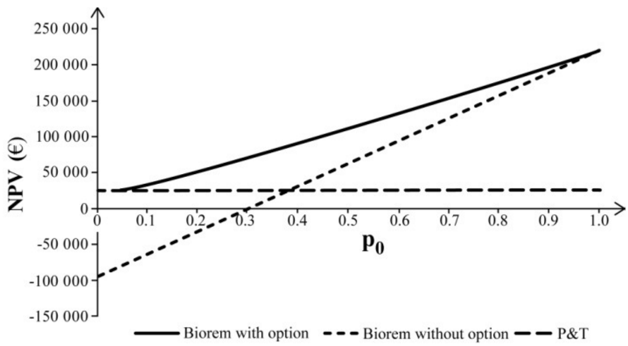


Figure 5.8 Impact of variations in p_0 on the NPV of the biostimulation strategy with and without option to abandon.

5.3.4 Lessons learned

This case study shows how a firm can deal with an uncertain contaminant mass removal efficiency due to variations in biophysical processes. It is demonstrated that if a firm does not have the option to abandon biostimulation if it proves to be inefficient, the firm will not adopt biostimulation, unless the initial belief in an efficient biostimulation strategy is sufficiently high (0.39 for this case study).

Further, it is shown that if the cost required to redirect the remediation strategy is high, the firm is not likely to use the option to abandon. Also the value of the remediation is an important element within the analysis. The sensitivity analysis for this value allows the firm to make a deliberate decision on the adoption of biostimulation or pump & treat.

The bioremediation case study demonstrates how the integration of the real options theory in the investment decision can support the development of this new technology. It indicates under which conditions this remediation strategy should continue operating or when further research is required.

5.4 The option to abandon: phytoremediation

5.4.1 Site description

The site considered is a former military landing field at which 100 000 liters of kerosene leaked in the soil due to an accidental spill-over. Initial maximum concentrations mineral oil equaled 3 500 mg/kg. As a result, an area of 4 650 m^2 was excavated to a depth of about 7 m. The excavated soil was partly considered clean. To remediate the contaminated volume (about 16 320m^3), biopiles are constructed (a general description on the application of biopiles is presented in Annex I of this chapter). The contaminated soil is first sieved, nutrients are added and then the soil is put in piles. After 17 months or 1.42 years, the soil is considered clean.

The groundwater table is situated at a depth of 8 m below ground surface. Initial maximum concentration mineral oil equaled 350 000 µg L^{-1}. Groundwater is not remediated but monitored during a period of 14 years.

5.4.2 Results of the real options model: a phytoremediation case study

Table 5.1 gives an overview of the costs associated with excavation and phytoremediation. For the excavation strategy, the investment costs includes the cost for the excavation, construction costs of biopiles, the installation of monitoring wells and monitoring activities during the excavation and construction of biopiles. Also the reports concerning the follow-up of the remedial works are included. The annual cost is the cost for the monitoring of groundwater quality. The investment cost of the phytoremediation strategy includes the planting of trees, the installation of monitoring wells and the cutting of trees after clean-up. Annual costs include mowing and monitoring.

Table 5.1 Cost (1000€) for the excavation and phytoremediation strategy

Excavation strategy (Exc)		Phytoremediation strategy (Phyto)	
Excavation (I_{exc})	412.3	Planting of trees (I_{phyto})	6.3
Biopiles (I_{exc})	110.0	Monitoring wells (I_{phyto})	11.0
Soil disposal (I_{exc})	272.4	Cutting trees after clean-up	12.9
Monitoring excavation (I_{exc})	121.5	Annual mowing cost	1.0
Monitoring wells (I_{exc})	9.6	Annual groundwater monitoring	5.2
Annual groundwater monitoring	5.2	Annual soil monitoring	1.9

After the soil is excavated, annual monitoring of groundwater quality is required during 10 years (from year 2 until year 12). After less than two years, the soil was considered clean. Hence, at that time, the property can be sold. The value of the remediation is set at €23 m^{-2} which corresponds to current property prices for an industrial area. The revenue from the property sale (R) equals €106 904. To calculate the NPV for the excavations strategy, the cost data presented in Table 5.1 is applied. A discount rate of 0.05 is used. The NPV of the excavation (Exc) strategy equals

$$\mathrm{NPV_{Exc}} = \frac{\mathrm{R}}{(1+\mathrm{r})^{\mathrm{T}}} - \sum_{t=2}^{12} \frac{\text{monitoring cost}}{(1+\mathrm{r})^{\mathrm{t}}} - \mathrm{I_{Exc}} = -\text{€}866\,951,$$

For the phytoremediation strategy, the annual quantity of contaminated mass removed can vary between 6.77 and 10.71 ton, leading to a 25 and 5 year remediation period (T) respectively. The annual cost includes both soil monitoring, groundwater monitoring and annual mowing costs. The soil is monitored until clean-up is achieved. Groundwater is assumed to be monitored at least 14 years, also if soil clean-up is achieved more early. If it takes more than 14 years to achieve soil clean-up, groundwater (GW) monitoring continues as well. The investment cost includes the installation of monitoring wells and the planting of trees. The NPV is calculated as follows:

$$\mathrm{NPV_{Phyto}} = \frac{\mathrm{R} - \mathrm{C_{cutting\,trees}}}{(1+\mathrm{r})^{\mathrm{T}}} - \sum_{t=0}^{\mathrm{T_{GW}}} \frac{\text{GW monitoring}}{(1+\mathrm{r})^{\mathrm{t}}} - \sum_{t=0}^{\mathrm{T}} \frac{\text{soil monitoring}}{(1+\mathrm{r})^{\mathrm{t}}} - \mathrm{I_{Phyto}},$$

The NPV of phytoremediation is calculated based on the cost data presented in Table 5.1. T_{GW} represents the period during which groundwater quality is monitored (T_{GW} varies between 14 and 24 years). Soil quality is monitored during the remediation period (T varies between 5 and 25 years). The revenue (R) from the property sales equals €106 904. Because the groundwater table is situated 7m below ground surface, the contamination does not pose any threat to human health. Therefore, it is assumed that the property can be sold from the moment that the soil is considered clean. When a discount rate of 0.05 is applied the NPV equals -€25 281 and -€128 644 for a 5 year and 25 year remediation period respectively. This result makes clear that despite an uncertain remediation time, the decision maker should invest in the phytoremediation strategy.

However, it is possible that the decision maker not only considers the revenue of the property sale as the benefit of the remediation. Also the value of a clean site (Cf. Chapter 2) or the value of redevelopment at the area can be considered as the benefit of the remediation. The value of redevelopment can involve the profit a firm obtains from selling houses, renting offices, or profit resulting from industrial activity. Because this value is unknown, the real options theory is applied to determine the value at which excavation becomes a relevant remediation strategy. Table 5.2 shows the value of the different parameters. The quarterly net cash flow of an efficient and inefficient phytoremediation strategy are presented by a and b respectively.

Table 5.2 Parameter values for the real options model, assuming a property value of €23 m^{-2}

Parameter	Value
a	-€101
b	-€1 331
λ	0.7
p_0	0.7
r	0.0123
NPV Exc	-€866 951

Figure 5.9 shows that if the property value increases to €1 150 m^{-2}, the decision maker will never adopt phytoremediation. If the value of the property is for instance €500 m^{-2}, a firm will adopt the excavation strategy if 5 bad samples in excess of good samples are taken. For a property value smaller than €280 m^{-2}, the decision maker should never adopt the excavation strategy. The excavation strategy never dominates.

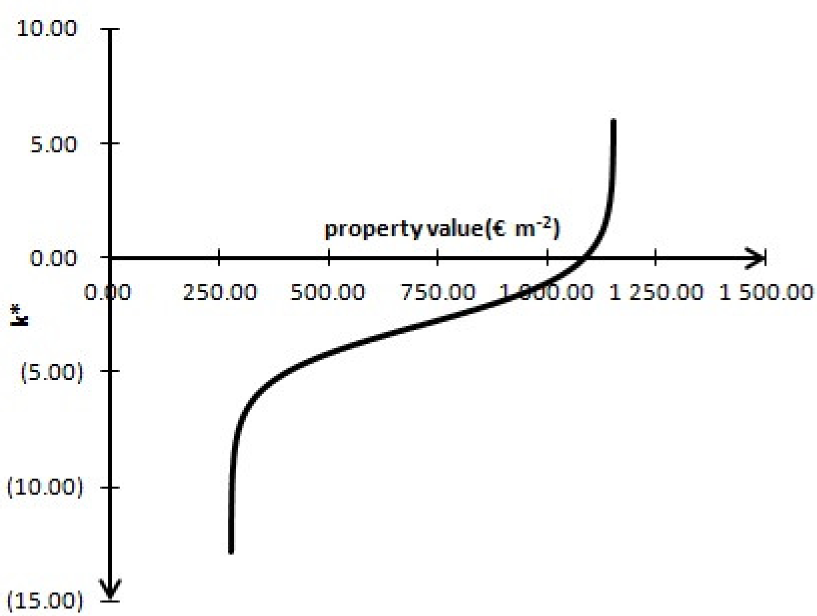


Figure 5.9 k* and p* in function of the property value (€ m^{-2})

If the value of the property is set at €500 m^{-2}, k* equals -4.18 which is also shown in Figure 5.10. The value of p* equals 0.06. When 5 bad samples in excess of good samples are taken, the decision maker will abandon the phytoremediation strategy and start excavating. The critical belief in an efficient phytoremediation strategy is decreased from 0.7 to 0.06.

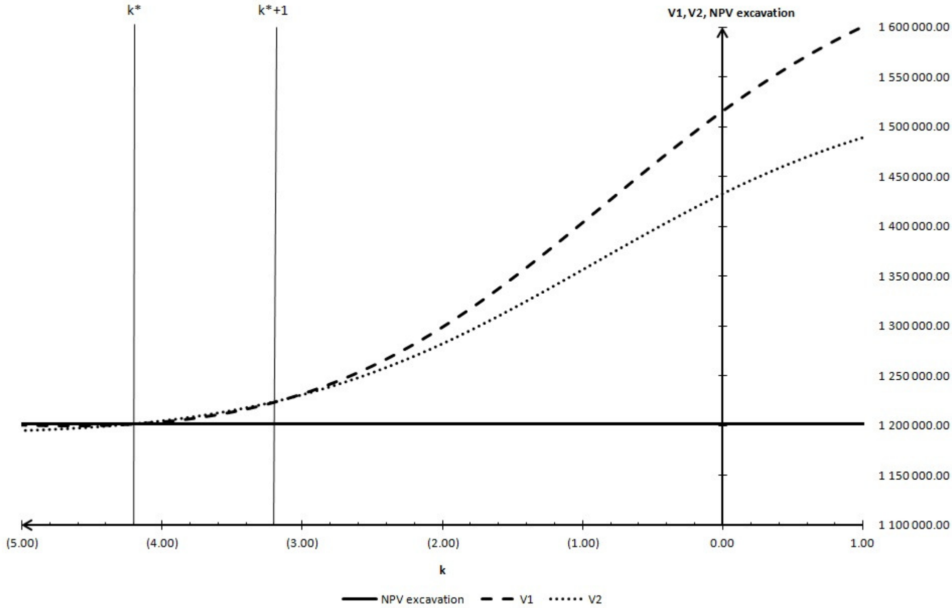


Figure 5.10 V1, V2 and NPV excavation in function of k

Figure 5.11 shows the impact of changes in the value of λ on k* and p* for a property value of €500 m^{-2}. If λ equals 0.5, the samples do not provide any information and the decision maker will never adopt the excavation strategy. The critical belief in an efficient remediation strategy is then about 0.57. The higher the value of λ, the less bad samples in excess of good samples are required to adopt the excavation strategy and the more the decision maker will be certain that the phytoremediation strategy is inefficient.

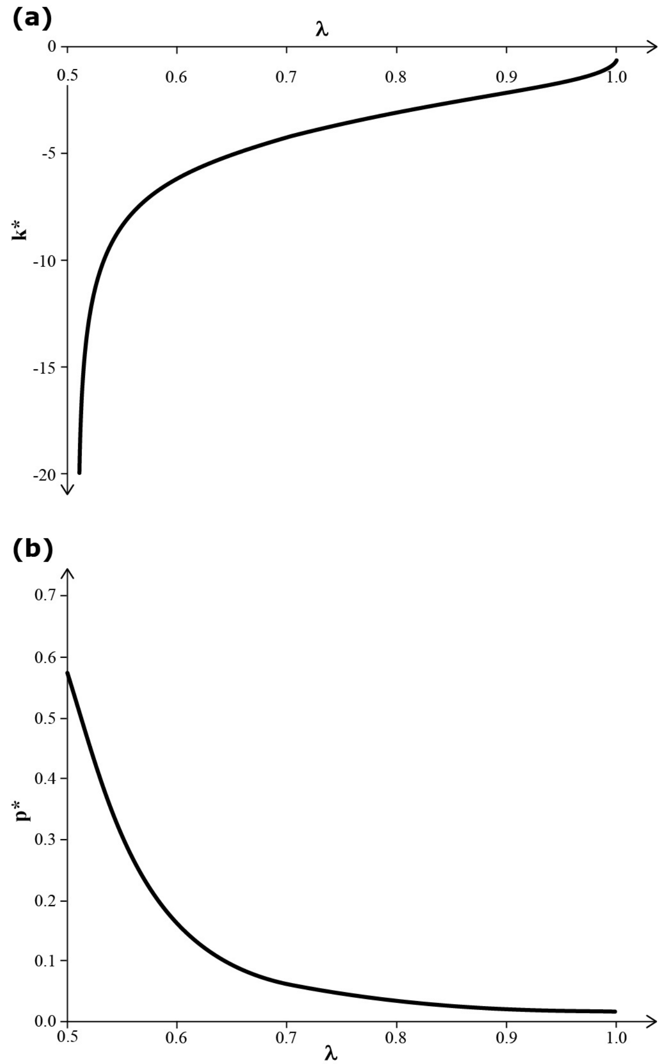


Figure 5.11 k* and p* in function of λ

For the base case scenario, an efficient phytoremediation strategy is defined by setting the remediation time at 5 years (referred to as the lower limit). Figure 5.12 shows the effect on k* when the remediation time for an efficient strategy increases. The less efficient phytoremediation is, the less bad samples in excess of good samples are required to redirect the remediation strategy. If the remediation time would vary between 12 and 25 years the decision maker will never adopt the phytoremediation strategy.

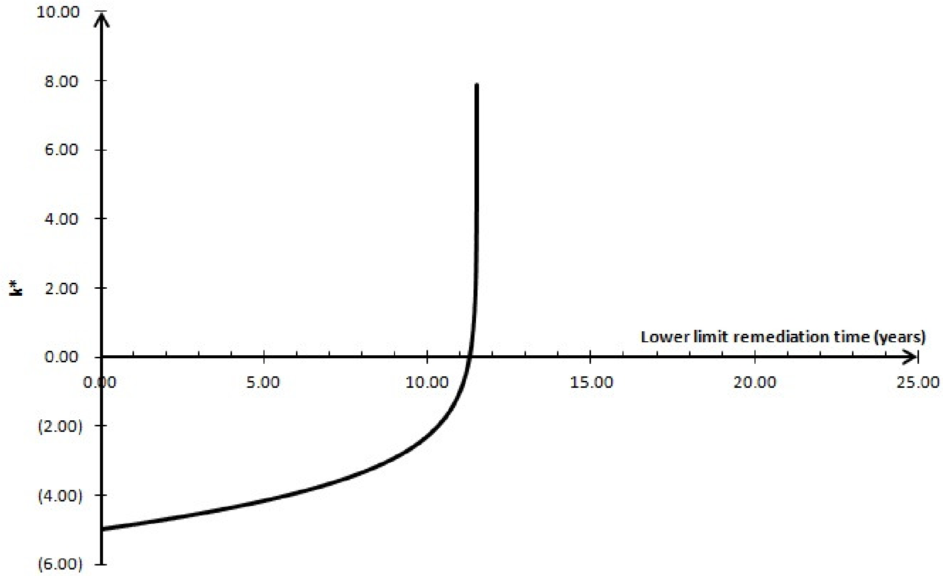


Figure 5.12 The effect of a less efficient phytoremediation strategy on k*

5.4.3 Lessons learned

To clean up soil contamination, excavation is an often applied remediation strategy. This case study shows that if only the sales value of the property is considered, a firm should never adopt excavation, even if the application of phytoremediation takes 25 years to achieve full clean-up. However, a firm that would like to redevelop the contaminated land, may wish to keep the remediation time as short as possible so as to earn revenues from the redevelopment. In that case, the value of a remediated site increases, and excavation can become an economically justifiable remediation strategy.

This case study shows that if redevelopment increases the property value to €500 m^{-2}, phytoremediation is not necessarily the preferred remediation strategy. If the remediation time of phytoremediation is more than 12 years, the firm should excavate the contaminated site. Whether €500 m^{-2} is a realistic value depends on the kind of activity that is developed and on which level the evaluation is made. Roughly spoken, for an area of 5 000 m², a property value increase of €477 m^{-2} corresponds to a profit margin of €2 385 000. On a micro-economic (company) level, this seems a large value. On a macro-economic (society) level, this value can also include the value of extra employment and increased consumption.

5.5 Conclusion and discussion

This study considers the investment in a phytoremediation and biostimulation strategy under technological uncertainty as a reversible decision. Technological uncertainty is gradually resolved in a Bayesian process by evaluating the project at determined points in time. During the time the remediation strategy is in operation, one can observe its efficiency and hence learn from its application. It is demonstrated how the integration of the real options theory in the investment decision can support the development of gentle remediation strategies. It indicates under which conditions these strategies should be continued or when further research on their effectiveness is required.

The case studies show that also the correctness of the samples taken highly influences the decision making process. However, in reality, the distinction between a correct and an incorrect sample may be less straight forward than assumed in the real options analysis.

Also note that for the case study considered, it is assumed that contaminant mass removal continues indefinitely. Also the quantity of contaminant mass already removed by the biostimulation strategy before the decision to redirect the remediation strategy is not taken into account. Nevertheless, this study demonstrates how the integration of the real options theory in the investment decision can support the development of innovative technologies.

The real options theory integrates more aspects related to site remediation than the calculation of the cost effectiveness of different remediation strategies. Besides the technical, hydrogeological and economic aspects, also groundwater sampling and the confidence a firm has in the application of a particular remediation strategy is taken into account. In that way, the use of the real options theory reveals more information that forms the basis for substantiated and justifiable decision making. For this reason, policy makers should integrate the real options analysis in the existing evaluation tools. The use of the real options theory will stimulate experimentation with innovative remediation strategies. As a consequence, remediation experts will learn about the merits of phytoremediation and bioremediation which can increase future applications.

5.6 Annex I The use of biopiles as a soil remediation strategy

Treatment by biopiles can involve both contaminant reduction by aeration for volatile organic compounds (gasoline) and biodegradation for midrange products (diesel, kerosene) that contain lower amounts of volatile components. A biopiles treatment involves the excavation of contaminated soil which is aerated and enriched by minerals and nutrients to simulate biodegradation. The soil is piled into heaps which have an underground system through which air passes and which may be covered to prevent run-off, evaporation and volatilization [108, 109]. If volatile organic compounds are present, the air may need to be treated before being discharged into the atmosphere. Soil texture, permeability, and moisture content play a major role in the success of biopiles. Low permeable soils are difficult to aerate and often clump together, making the distribution of air and nutrients difficult. Turning of the soil may be required to promote optimal conditions for biodegradation [109, 110].

6 Conclusion and discussion

6.1 Conclusion

6.1.1 How to determine the cost effectiveness of phytoremediation and bioremediation?

The aim of this dissertation is to obtain a correct economic evaluation of the cost effectiveness of phytoremediation and bioremediation in comparison to other remediation technologies. A cost effectiveness analysis can be performed on an average or incremental basis and different units of the effect can be considered. The use of these different calculation methods results in different conclusions. A final conclusion depends on how the decision maker prefers to express the effects and how these effects are valued.

This study provides an extended insight into the different calculations of cost effectiveness. It is shown that when besides the average cost effectiveness ratio (ACER), also the incremental cost effectiveness ratio (ICER) is calculated, the decision maker is better informed to select a remediation scenario. The ACER determines the average cost per effect (C/E). The ICER is calculated as the ratio of the change in cost to the change in effect, *i.e.* $\Delta C/\Delta E$. Although a definite decision based on the ICER can only be made when the willingness-to-pay for an incremental effect is known, the ICER better defines the trade-off between costs and effects compared to the ACER and a link to a cost benefit analysis is provided. The presented methods can be generally applied at other contaminated sites to evaluate whether phytoremediation or bioremediation are valuable options.

6.1.2 Are phytoremediation and bioremediation cost effective groundwater remediation strategies?

A few conditions are to be primarily fulfilled to successfully apply phytoremediation and bioremediation. As regards phytoremediation, it is not only required to have the necessary space for planting the trees, also root proximity, plant tolerance to the contamination and a sufficient growth rate are key elements as to whether phytoremediation can be considered as a viable option or not [11-18]. Moreover, the phytoremediation and bioremediation strategies considered in this dissertation rely on microbial degradation processes to remediate soil and groundwater contamination. Factors such as strain selection, microbial ecology, the type of contaminant, and environmental constraints, can lead to variations in biodegradation and hence, are crucial for successful clean-up [24, 25, 50]. For the case studies examined, it is assumed that the primarily conditions are fulfilled.

The selection of a remediation strategy is always site specific. Besides hydrogeological site conditions, also contaminant characteristics, technological aspects, cost data and not at the least the value of remediation are important elements that should be taken into account during the decision process. For each location, these aspects differ which makes it impossible to draw general conclusions on the cost effectiveness of phytoremediation and biostimulation. The result of this dissertation is an extended insight in the use of a cost effectiveness analysis as an economic evaluation tool that supports decision making in soil and groundwater remediation.

Chapter 3 demonstrates that both phytoremediation and biostimulation can be considered as valuable alternatives among the remediation strategies available. Whether phytoremediation or bioremediation should be selected or not, depends on the benefit of the remediation. In section 3.1.4, it is shown that if the remediation strategy’s objective is to prevent illness, natural attenuation is the preferred remediation strategy. Concentrations are low and hence, health risks are limited. However, if the objective is to prevent the plume from further spreading, results can be different depending on the value of the damage incurred. The contaminated groundwater can reach surface water, a groundwater extraction well or cause damage to adjacent sites.

Further, section 3.1.5 demonstrates that if the area where concentrations are measured is located closer to the source zone, natural attenuation does not achieve to prevent the plume from further spreading, while phytoremediation certainly does. Moreover, when the remediation strategy at the source zone is out of operation (section 3.1.6), the phytoremediation strategy also prevents the contamination from spreading further. In both cases, the reason that phytoremediation achieves the objective is the biodegradation process. Whether phytoremediation should be selected or not, depends on the value of the remediation.

Concerning biostimulation a similar conclusion can be drawn. The average cost effectiveness ratio demonstrate that the adoption of biostimulation is cost effective. The calculation of the ICER leads to a more nuanced conclusion, biostimulation should only be selected if, compared to a containment strategy, the value of the incremental effect is worth the additional costs.

6.1.3 How important is the use of hydrogeological groundwater modeling?

The phytoremediation case study presented in Chapter 3 clearly demonstrates the importance of hydrogeological modeling. The results of this model show the impact of remediation technologies on groundwater flow and contaminant concentrations. Moreover, the groundwater model shows that the groundwater extraction by poplar trees does not affect groundwater flow at the site under consideration. Besides, also groundwater extraction at the source zone of that site does not lead to a cone of depression and hence, does not prevent the contamination from moving towards the border of the site.

Furthermore, it is determined that when the design of a remediation strategy is only based on rough-and-ready rules, not taking into account the interaction of the different hydrogeological aspects and their effect on groundwater contamination, groundwater remediation can become needlessly costly. this analysis does not only demonstrate that the correct use of groundwater flow and transport models can lead to better decision-making resulting in the application of a more effective and less expensive groundwater remediation strategy. It is also shown how valuable a model – and the advice of independent researchers – can be.

6.1.4 How to integrate technical uncertainty in the decision making process?

Although the cost effectiveness analysis achieves at providing a concise representation of key decision making issues, it does not consider technical uncertainty. To integrate the concept of technical uncertainty and decision reversibility into the decision analysis, first a decision tree is developed. This decision tree demonstrates the existence of an option value associated with the possibility to redirect the remediation strategy. It is demonstrated that the value of the remediation strategy increases when it is made possible to conduct the remediation in different stages.

When the decision tree is applied, the decision maker has the possibility to redirect the remediation strategy, but only at one point in time. However, since multiple samples can be taken during a remediation processes, the parties responsible for remediation have the option to redirect the remediation strategy at any point in time. Therefore, a real options model that defines the optimal timing of redirecting a remediation strategy is developed.

This model considers the investment in a phytoremediation or bioremediation project under technical uncertainty as a reversible decision. Technical uncertainty is gradually resolved in a Bayesian process by evaluating the project at determined points in time. For the bioremediation case study, it is demonstrated that when this option is taken into account, the decision maker decides to invest in the bioremediation strategy, while at the same time it determines an optimal timing to abandon bioremediation if its operation proves to be inefficient. It is shown how the belief in an efficient bioremediation strategy is updated based on the samples taken. Also the importance of the probability with which the samples reflect the true state of the world, is demonstrated.

6.1.5 What is the optimal timing to abandon the applied remediation strategy if it fails to remove the contamination sufficiently?

Chapter 4 and 5 demonstrate that due to the variability in biodegradation rates and hence, technical uncertainty, there exist a value in the option to redirect the remediation strategy. It is shown that when the decision maker has the option to abandon phytoremediation or bioremediation when the application proves to be inefficient, the value of the remediation increases. As regards biostimulation, its costs are relatively high compared to phytoremediation which makes it necessary to have the option to abandon. For the given remediation value, the firm would only invest in bioremediation if the option to abandon is considered.

Concerning phytoremediation, the costs for excavation is more than 10 times higher and when only the property value is considered as the benefit of the remediation, phytoremediation is preferred even without the option to abandon. The variability in the bio-physical processes do not have an impact on that result.

For both cases, it is indicated under which conditions biostimulation and phytoremediation can continue operating or when further research is required. A sensitivity analysis on the property value and the range in effectiveness indicate for which values the more conventional remediation strategies dominate.

This study can be extended to the adoption of other innovative technologies that are subject to technical uncertainty. The literature on technology adoption mostly considers the *ex ante* adoption problem under technical uncertainty taking into account the option value of waiting [111]. Often, it is find that the green investments' irreversibility and uncertainty about related benefits might delay environmental innovations which can enforce lock-in of currently applied technologies [112-114]. This real options analysis not only considers the *ex ante* decision analysis of the investment in a new technology under uncertainty, it also allows for an *ex post* evaluation of the investment. The decision maker includes the possibility to evaluate the performance of the new technology into the decision making process. By giving firms the possibility to switch to more efficient technologies if necessary, experimentation with alternative technologies is encouraged, which can lead to technological change.

6.2 Discussion

6.2.1 Cost effectiveness

The cost effectiveness analysis is only one step in the approach to groundwater contamination. Before the cost effectiveness for the different remediation scenarios is evaluated, a remedial objective is set. However, because the risks posed by the spreading of the contamination are not clearly assessed, the benefits of achieving the objective are not known. It is assumed that achieving the objective is worth the costs. To justify the remediation cost, further evaluation of the remediation benefits is necessary.

Furthermore, to calculate the cost effectiveness of different remediation strategies, only the private costs to the firm are considered and the effect is determined in contaminant mass removed or remediation time only. The remediation strategy itself can also have an impact on the environment and its ecosystems. Therefore, social costs and benefits should be taken into account for CEA to become an appropriate guide for a more efficient allocation of resources. The inclusion of these impacts can alter the ranking of remediation alternatives and can increase the magnitude of the ratios.

Also note that only the costs are discounted when the cost effectiveness ratio is calculated. In that way, it is assumed that all the benefits occur at once, in year 0 and not gradually over time. Discounting the effectiveness of different remediation strategies will lower total effectiveness, leading to a larger cost effectiveness ratio. This subject leads to the discussion of discounting environmental effects. Each positive discount rate makes the value of future environmental effects shrink, leading to modest actions now.

6.2.2 Dealing with technical uncertainty

This dissertation points out the existence of variation in biophysical processes, leading to uncertainty in contaminant mass removal efficiency. However, a probability density function, defining the variation of these bio-physical processes is not determined. Only a range between which the biodegradation rate varies, is used to analyze the decision tree and to develop the real options framework.

As regards the decision tree, the probabilities attached to each contingency are selected to exemplify the outcome of a decision tree. A sensitivity analysis demonstrates the impact of changes in these assumptions on the outcome. For a real-life case study however, these probabilities should be based on historically observed frequencies or subjective assessments by experts based on information or theory [63].

Also note that in Chapter 4 and 5, the application of P&T is considered as a remediation strategy with a certain removal efficiency. However, also P&T systems can fail to meet the specified objectives, due to an inappropriate design or unexpected rebound effects [10]. Therefore, it would be useful to include uncertainty inherent to the application of P&T into the economic analysis as well. Moreover, it is assumed that contaminant mass removal continues indefinitely. Also the quantity of contaminant mass already removed by the biostimulation strategy before the decision to redirect the remediation strategy is not taken into account.

Nevertheless, this study demonstrates how the integration of the real options theory in the investment decision can support the development of innovative clean technologies. This analysis should be considered as a tool to spur the escape from technological lock-in and to keep different technological options open. Moreover, this tool not only supports decision making for groundwater remediation, it can also be used by governmental authorities that aim to increase investments in environmental friendly technologies. By giving firms the possibility to switch to more efficient technologies if necessary, experimentation with alternative technologies is encouraged, which can lead to technological change.

6.2.3 Groundwater modeling

A deterministic groundwater flow model is used to design the different remediation strategies and to evaluate how effective each remediation scenario is. This model however, does not address uncertainties regarding the effectiveness. In order to quantify the uncertainties associated with subsurface heterogeneity and to address the dynamics of flow and transport, stochastic groundwater modeling should be applied. Also note that the model results are produced for each discrete year. Regarding the phytoremediation case study (Chapter 3), alternatives P&T and phytoremediation both reach the 150 mg L^{-1} objective after one year but the concentrations achieved are much lower when P&T is selected than when phytoremediation is adopted. Hence, for the P&T remediation strategy, the objective is reached within a year.

Further, the complexity and uncertainty among biological processes associated with bioremediation and phytoremediation are not modeled. However, the variation in biodegradation rates applied as input in the hydrogeological simulation can be considered as the output of models simulating these biological processes. The different biodegradation rates applied in the development of the decision tree can be attached to different sorts of contamination.

6.3 Recommendations to policy makers and remediation experts

The procedures that exist nowadays to select an appropriate remediation strategy are overly simplified. Costs and effects are evaluated on a scale from 1 to 10, a true cost effectiveness analysis is never applied. Policy makers should develop a decision tool that allows for a correct comparison of cost and effects. Furthermore, the effectiveness of groundwater remediation strategies should be based on the results of a groundwater model, even though site characterization and the development of a groundwater model is time consuming and costly. Also uncertainties should be integrated in the economic analysis. The real options analysis is a structured and rationally justifiable method that supports decision making under uncertainty. It stimulates experimentation with innovative remediation strategies. As a consequence, remediation experts can learn about the merits of this remediation technologies which can increase future applications. The use of both the cost effectiveness analysis and the real options analysis leads to informed decision making that encounters the complexities inherent to remediation problems.

To make informed decisions, remediation experts and firms responsible for clean-up should broaden their expertise in a variety of remediation technologies. Currently, the knowledge on phytoremediation and bioremediation is limited and concentrated among a few scientists. In order to stimulate the adoption of phytoremediation and bioremediation, remediation experts or firms responsible for clean-up should learn about their merits. Moreover, remediation experts should apply hydrogeological modeling and profound economic evaluation methods to select the appropriate remediation strategy. Their application results in a better understanding of uncertain parameters, a reduction in remediation time and cost savings.

6.4 Questions for further research

6.4.1 Valuing groundwater remediation

The results of the economic evaluation indicate the importance of the remediation value. To make case specific decisions, it is necessary to determine the value of the remediation by using stated or revealed preference methodologies.

Also the external costs and benefits of remediation technologies should be determined and integrated in the economic analysis. Phytoremediation can be considered as an environmental friendly remediation strategy that preserves soil fertility and structure. The application of excavation and pump & treat can cause damage to the soil structure, the hydrogeological cycle and ecosystems. The *ex situ* treatment of soil and groundwater not only involves energy use, the use of granular activated carbon results in waste products that should be disposed of as well. The internalization of the external effects affects decision making as it changes the cost structure.

6.4.2 Real options

Concerning the real options analysis, only technical uncertainty is considered. Although this dissertation addresses hydrogeological and economic uncertainties in a sensitivity analysis, these uncertainties can also be integrated in a real options analysis. Hydrogeological parameters are often not sufficiently specified and can spatially vary within the same site. Collecting more information on site specific conditions can lead to a more effective remediation design and potential cost reductions. The revenues resulting from the redevelopment of contaminated sites is another aspect that should be integrated in the decision analysis. A firm will not invest in marginal land or brownfields if the planned economic activities do not lead to sufficient incoming cash flows. The real options theory can be applied to determine the optimal timing to start the remediation activities. It can be determined how many tests are to be conducted in order to design an effective remediation strategy or, the level at which the market value justifies remediation costs, can be defined.

The case studies considered in this dissertation only involve industrial sites. The real option theory can also be applied to economically evaluate remediation technologies for contaminated residential areas, or agricultural land. For instance in the Campine region, many hectares of farmland are contaminated with heavy metals at which only fodder-plants can be grown. By comparing the income of a farmer who has the option to choose among different crops to cultivate and a farmer that can only grow fodder-plants, it can be determined whether remediation of those fields is economically justified.

To make the economic analysis more realistic, it is necessary to take into account the quantity of contaminant mass already removed by the remediation strategy initially applied. Also a boundary condition that indicates that total contaminant mass removal is achieved should be included. To get more insight into the critical aspects that stimulate investments in gentle remediation technologies, further research can examine the extent to which adoption and diffusion of gentle remediation technologies are stimulated by integrating the option to abandon.

6.4.3 Combining remediation strategies

Both the cost effectiveness and real options analysis focus on the comparison of remediation strategies, but their combined application should be investigated as well. It should be evaluated to which extent each remediation technology can deal with the risks assessed. Contaminated sites are often complex regarding the different types of pollutants present and heterogeneity of the subsurface. It is required to investigate whether phytoremediation, excavation and pump & treat remediation strategies can deal with this kind of complexities separately. Maybe, it is more economically feasible to remediate these kind of sites using a combined approach. This approach involves a clear examination of the extent to which source zones should be excavated, the contaminants that are tolerated by phytoremediation, and the timing of the redevelopment of the different locations.

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