COMPUTATIONAL TOOLS FOR IMPROVED ANALYSIS AND ASSESSMENT OF GROUNDWATER REMEDIATION SITES

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ABSTRACT

Remediation of contaminated groundwater remains a high-priority national goal in the United States. Water is essential to life, and new sources of water are needed for an expanding population. Groundwater remediation remains a significant technical challenge despite decades of research into this field. New approaches are needed to address the most severely-polluted aquifers, and cost-effective solutions are required to meet remediation objectives that protect human health and the environment.

Source reduction combined with *Monitored Natural Attenuation* (MNA) is a remediation strategy whereby the source of contamination is aggressively treated or removed and the residual groundwater plume depletes due to natural processes in the subsurface. The USEPA requires long-term performance monitoring of groundwater at MNA sites over the remediation timeframe, which often takes decades to complete. Presently, computational tools are lacking to adequately integrate source remediation with economic models. Furthermore, no framework has been developed to highlight the tradeoff between the degree of remediation versus the level of benefit within a cost structure.

Using the Natural Attenuation Software (NAS) package developed at Virginia Tech, a set of formulae have been developed for calculating the TOR for petroleum-contaminated aquifers (specifically tracking benzene and MTBE) through statistical techniques. With the knowledge of source area residual saturation, groundwater velocity, and contaminant plume source length, the time to remediate a site contaminated with either benzene or MTBE can be determined across a range of regulatory maximum contaminant levels.

After developing formulae for TOR, an integrated and interactive decision tool for framing the decision analysis component of the remediation problem was developed. While MNA can be a stand-alone groundwater remediation technology, significant benefits may be realized by layering a more traditional source zone remedial technique with MNA. Excavation and soil vapor extraction when applied to the front end of a remedial action plan can decrease the amount of time to remediation and while generally more expensive than an MNA-only approach, may accrue long-term economic advantages that would otherwise be foregone.

The value of these research components can be realized within the engineering and science communities, as well as through government, business and industry, and communities where groundwater contamination and remediation are of issue. Together, these tools constitute the **S▪E▪E▪P▪AGE** paradigm, founded upon the concept of *sound science for an environmental engineering, effectual economics, and public policy agenda.* The TOR formulation simplifies the inputs necessary to determine the number of years that an MNA strategy will require before project closure and thus reduces the specialized skills and training required to perform a numerical analysis that for one set of conditions could require many hours of simulation time. The economic decision tool, that utilizes a life cycle model to evaluate a set of feasible alternatives, highlights the tradeoffs between time and economics can be realized over the lifetime of the remedial project.

DEDICATION

To my father, role model, mentor, and inspiration:

Joshua Allen Joseph, Sr.

In Memoriam

Dad, there is no way I could have made it through this without you. Although you departed this world on June 19, 1989, you have never left my heart, my thoughts, or my prayers.

Little did I know the role your calculus-saving advice would play in my life, well beyond my freshman year at our Alma Mater. How monumental were the words – *the graph of a function never changes*. It did not fully sink in then, but I know you know now how those words truly changed my life.

I did the best that I could to honor my promise
In what was one of the last heart-to-heart conversations we shared…

When you and I walk across the Virginia Tech commencement stage, we will walk together for you have spoken to me every day of this journey, loudest when I needed to hear it strong and softest when I needed to hear it easy.

There is no doubt that my drive to be the best comes from you… I love you and thank you for all that you have done and continue to do for me while on this journey.

Always,

Your Son!

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Then came the children of Israel, even *the whole congregation, in to the desert of Zin…*
And there was no water for the congregation…
…
And Moses took the rod from before the Lord, as he commanded him.
…
And Moses lifted up his hand, and with his rod he smote the rock twice: and the water came out abundantly, and the congregation drank, and their beasts also.

Numbers 20: 1, 2, 9, 11
(*The birth of Groundwater Hydrology*)

I am thankful to God and to my Lord and Savior, Jesus Christ, for granting me the capacity to weather the storm and persevere through the trials and tribulations that are the burden, the essence, and the challenge of pursing the doctorate of philosophy. It has not been an easy journey; however, through these series of life-defining minutes, moments and measures, I have been forever and fundamentally meld into a new creature, a new engineer, a new scholar.

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CHAPTER ONE — INTRODUCTION

Environmental site remediation transpires because remedial actions are required by governing authorities. Contaminants in the air, water or earth are mitigated because it is *the law*. At face value, it seems simple enough. There is a requirement to clean up; therefore, sites are cleaned up. Unfortunately, the satisfaction of this mandate is anything but simple. And often, the complexities and controversy continue to escalate once the directive has been issued. The edict has probably resulted from a significant amount of investigation into some compound existing in some media that causes some harm to one or more receptors. This research focuses exclusively on the difficulties of cleaning up groundwater at sites contaminated with light nonaqueous phase liquids, specifically, benzene and methyl tert butyl ether.

The cleanup processes are often quite expensive and very time intensive. At many sites, natural attenuation has proven to be a significant remedial technique for addressing dissolved petroleum hydrocarbon contaminants. There is also evidence that reaching site remediation goals via natural attenuation processes may provide a significant cost savings than through the use of conventional remediation technologies. Furthermore, the incorporation of risk-based remedial objectives is considered especially beneficial in producing more timely and cost-effective solutions to site cleanup goals.

This dissertation details the impetus, methodologies, results and conclusions that have resulted in (1) the formulation of a procedure for calculating the time of remediation for a petroleum-contaminated aquifer, (2) the formulation of an integrated framework and interactive decision analysis tool for combined engineering/economic evaluation, and (3) the use of an in-field case study to test and assess the implementation of the engineering and economic tenets within the **S▪E▪E▪P▪AGE** paradigm – *sound science for an environmental engineering, effectual economics, and public policy agenda.*

1.1 BACKGROUND

The simple statement that *water is essential* implies that it must be made available. *People* are dependent upon water. We drink it. It is an essential component of our domestic sewerage

transport and storage. We need water for our basic hygienic needs and we need it as we perform the most basic of domestic tasks. As we continue to measure the *prosperity* of our society using the textbook economic indicators, the true health of our society can be closely related to our capacity to provide for the basic needs of its members. No need is more basic than water.

According to the United States Geological Survey (USGS) (Waller, 1988), our nation's groundwater use increased by 52 billion gallons per day from 1950 to 1980. Furthermore, US EPA estimated that during the latter decades of this past century, groundwater was the source for approximately one-fourth of all fresh water use, which included approximately 35 and 95 percent of the drinking water supplied to urban and rural areas, respectively (US EPA, 1990a).

Groundwater, however, is more than a singular source of drinking water. It is relied upon, now more than ever (US EPA, 2004):

- Ground water supplies drinking water to half of the nation and virtually all people living in rural areas
- Ground water supports many billions of dollars worth of food production and industrial activity
- Ground water supplies the majority of streamflow in large areas of the country and provides much of the water in lakes and wetlands
- Ground water is used in every state and accounts for about one-quarter of all freshwater used in the United States today.

The imperative to protect and preserve groundwater is well documented in numerous textbooks (Bedient et al., 1997; Bedient and Huber, 1992; Viessman et al., 1989; Freeze and Cherry, 1979) that begin most introductions to hydrology with a discussion of the hydrologic cycle and the hydrologic budget, thereby introducing students to groundwater through hydrodynamic concepts. While this technical approach has been highly successful, as the field of hydrogeology has evolved, the textbooks have begun to introduce this specialized field by focusing on the essential and limited nature of this subsurface resource (Mays, 2005; Fetter, 1999; Fetter, 1988).

Freshwater constitutes only 3 percent of all the water on our planet with approximately one-third of this supply available to support the ecosphere (Hardisty and Özdemiroğlu, 2005) as approximately two-thirds of this fresh water supply exists outside of the glaciers and icecaps and constitute 95 percent of all fresh water resources (Joseph, 1995; Freeze and Cherry, 1979). These numbers are in the range of most reported quantities as evidenced by statistics from the Groundwater Foundation (2006) that state groundwater contributes 98 percent of the world's useable supply of available water.

Unfortunately, this resource is as threatened as it is limited. While the available data suggests that groundwater contaminant levels are usually below applicable drinking water standards, there are constant and continuous threats to the sustained quality of this resource. Groundwater quality is threatened by domestic, agricultural, industrial, commercial and waste disposal practices (US EPA, 1999b). The challenge is to address not only the source of this contamination, but also the transport pathways that lead from source to receptor. The ability and determination to meet these challenges are what has distinguished the technical progress and policy resolve of the United States from the nineteenth through the twentieth century and in comparison other nations in this century (Levin et al., 2002). Levin et al. (2002) assert that "these challenges are integrally interrelated, [and that an] integrated or at least systematic approach is necessary to facilitate efficient, effective, and sustainable solutions."

Data from the United States Environmental Protection Agency (US EPA) report that 245 groundwater-related disease outbreaks and 52,181 associated illnesses were reported between 1971 and 1985 (US EPA, 1990a). Before the 1970s, the common perception was that groundwater was naturally protected from contamination, due to the filtering processes of the geologic matrix. It is now widely held and accepted that groundwater contamination is a significant problem that must be addressed.

In *Ground Water Contamination: Transport and Remediation*, Bedient et al. (1997) state that the greatest potential source for ground water contamination is the massive amounts of organic compounds manufactured and used by industry, the federal government, agriculture, and municipalities. Benzene, toluene, ethyl benzene, and various isomers of xylene (BTEX) are of the group of soluble aromatic hydrocarbons that are commonly associated with petroleum fuels and

lubricants and cause for major concern at petroleum spill sites. These petroleum hydrocarbons (e.g., gasoline, diesel, motor oils, and similar products) can cause widespread damage across the environment as a result of spills or leaks. These contaminants may be present as a nonaqueous phase liquid or dissolved in the groundwater (US EPA, 1999b).

With respect to petroleum products, underground storage tanks had been the primary storage mechanism until the recent regulations that have seen an increase in above ground storage tanks. Early estimates were that approximately 5-6 million underground storage tanks (USTs) were in use across the country, storing gasoline, fuel, oil and other chemicals, with an estimated life span of just 18 years (US EPA, 1990a).

US EPA confirms that 447,233 releases from USTs occurred across the nation between 1988 and 2004. As of February, 2005, 412,567 cleanups had been initiated, 317,405 cleanups had been completed, and thus, 129,827 (or roughly 30 percent) were still incomplete. The Agency's goal of less than 10,000 new releases per year was met in 2004 as the number of new releases confirmed fell from 12,078 in 2003 to 7,848 (US EPA, 2005a). As comforting as this progress might appear, there is still a tremendous amount of progress that must be made as evidenced by the following data:

- Many of the 130,000 UST cleanups that remain in the national backlog are at difficult to remediate sites because of significant groundwater contamination
- Approximately 70 percent of states have reported that USTs are one of the major sources of potential groundwater pollution
- A general decrease in the number of cleanups (from 18,518 to 14,285) completed has been observed over the four year period ending in the year, 2004.

US EPA has developed six major UST cleanup initiatives to address the large number of cleanups that remain: (1) understanding the nature of sites, (2) targeting source water areas, (3) financing cleanups, (4) streamlining cleanups, (5) addressing methyl tertiary (tert) butyl ether (MTBE), and (6) recycling abandoned gas stations (US EPA, 2005a). These initiatives will depend upon systematic and integrated efforts across multiple disciplines and incorporating diverse scientists, engineers, and other professionals. As the Agency looks to mitigate the dangers posed to drinking water sources it is promoting collaborative efforts among state and regional drinking water resources and UST programs. In an effort to identify and fund fiscally sound financing options,

EPA is working with economists and project managers to develop implementation standards and tools for data reporting and analysis. The progress toward remediation solutions transpires concurrently with EPA's initiatives that promote new and innovative approaches to environmental regulatory control. In streamlining the cleanup process, the Agency is looking for states and other local entities to promote multi-site agreements and optimization plans that increase remedial effectiveness in a socio-politically and financially acceptable manner.

1.2 Problem Identification

Brady et al. (1998) has written that within the first third of this century the United States could spend anywhere between $400,000,000 and $1,700,000,000 to clean up hazardous waste sites, as mandated by the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), the Resources Conservation and Recovery Act (RCRA), and other federal requirements. Furthermore, old estimates from US EPA list approximately 2,000 CERCLA national priority list sites, between 1,500 and 3,000 RCRA sites, 7,300 Department of Defense (DoD) sites, 4,000 Department of Energy (DOE) sites, 350 other federal facilities, 20,000 state sites, and 295,000 sites leaking underground fuel from tanks.

The 2004 estimates that have been updated by US EPA suggest that site cleanup is still and will continue to be a problem. Due to the current regulatory requirements and practices, US EPA (2004) estimates that 125,000 underground storage tank (UST) sites will need to be cleaned up. This value, however, may be significantly underestimated as this number does not include sites where cleanup is complete or ongoing. Furthermore, this estimate is in recognition that several hundred thousand contaminated sites might not have yet been identified, for the majority of the hazardous waste sites identified were from four of the seven cleanup programs.

These UST sites provide an opportune springboard for innovative technology demonstration and assessment because they tend to be under the management of programs with relatively smaller and less complex cleanup projects versus the average programs administered by Superfund, DoD, and DOE (US EPA, 2004). UST sites account for a significant 43 percent of all cleanup sites. However, they comprise less than 10 percent of the estimated national cleanup costs, and are usually financed by property owners, state and local governments, and/or special trust funds that may have been based on dedicated taxes, such as fuel taxes. This niche environmental arena could

see cleanup costs reach close to $20 billion with an average per site cleanup of one-quarter of a million dollars.

While the prima facie conclusion is that contaminated sites require expeditious cleanup, a more expedient approach might compare this remediation cost to the value lost from spending what might otherwise have gone to benefit society through health care, education, social security, and infrastructure. Thus, the defense of these remedial activities becomes less credible as these expenditures fail to produce an equivalent value of human health and ecosystem protection (Brady et al., 1998). Milloy (1995) suggests that the actual health risk that is collectively posed by these contaminated sites is *relatively minor*, and that remediation of these sites will not return most of them to anything close to their preexisting condtion. The following statements, whether a source of concern or celebration, do begin to frame the increased search for and application of science-based, but cost-effective solutions to groundwater remediation problems:

From the National Research Council (1994):

> *It is now recognized that for many sites, there are few if any technical remedies for meeting the goal of a permanent remedy.*

From Freeze and Cheery (1989):

> *Attempts at aquifer remediation, where the goal has been to return a water supply to a state where its water meets drinking water standards, have almost without exception been a failure. If not an outright failure, they have made so little progress that exception is for success not to be achieved in this century [20th] and maybe not in the next one.*

From Brady et al. (1998):

> *In fact, drinking water standards are commonly the nominal, but unachievable cleanup target. It is therefore impossible to demonstrate that the billions of dollars being spent every year will improve human or environmental health, now or in the distant future.*

Monitored natural attenuation (MNA) has rapidly evolved throughout the 1990s and is now considered a viable and "cost-effective means of restoring contaminated ground water once immediate threats to public health from drinking contaminated ground water are not considered an environmental issue" (Sara, 2003). By 1995, natural attenuation was the second most popular option for sites with contaminated soils; it has become one of the most common treatment options at sites with contaminated groundwater. MNA is no longer viewed as the *do-nothing* or *walk-away* solution.

According to US EPA (1996) MNA may be deemed an appropriate remedial response given the demonstration of site-specific groundwater cleanup goals within a reasonable time frame (Sara, 2003).

In the situation where MNA with source reduction is under consideration as a remediation strategy to reach a cleanup standard, the critical issue is the time required to reach the cleanup goal (US EPA 1999a; US EPA 2005b). Chapelle et al. (2003) define time of remediation (TOR) at non-aqueous phase liquid (NAPL)-contaminated sites as the time it will take for a NAPL body of given mass and composition to dissolve and disperse to the point where predetermined water-quality standards are met at a site-specific point of compliance. While this definition is applicable to sites using standard-based remedial objectives, the TOR concept may be useful at sites where risk-based remedial objectives are formulated. Often this standard is the maximum contaminant level (MCL) for specific components of a fuel. Benzene and methyl tertiary butyl ether (MTBE) are often the contaminants of concern in setting and assessing remediation objectives but this criteria can vary from state to state and will depend upon the gasoline source. Because long-term performance monitoring is required to confirm progress toward achieving the remedial action objective at MNA sites (US EPA 1999a; US EPA 2005b), estimating TOR is important in the analysis and optimization of life-cycle costs and can influence future land use decisions, liability issues, and real estate transactions.

1.3 Research Objectives

A context for a relevant and novel research undertaking has been established. The priorities established by our national regulatory agenda are consistent with the concerns of citizens and stakeholders, alike: address and remediate sites of contamination in a manner that is protective of public and ecological health and that is cost effective for all affected entities. Through this multi-stage and interdisciplinary research the tenets of sound science, good public policy, and pragmatic engineered solutions are fused in an attempt to fashion a conglomerate set of tools that shape the decision making process relative to the remediation of contaminated groundwater sites. These tools are formed via the three major research objectives that follow:

1. A formulation for calculating the time of remediation for a petroleum-contaminated aquifer is developed, specifically tracking benzene and MTBE, using computer simulations and standard statistical techniques.

2. Applied science and engineering, economics, and social policy converge in the formulation of an integrated framework and interactive decision analysis tool that weighs the economic costs of multiple remedial alternatives as a prelude for determining the optimal remedial alternative.
3. The **S▪E▪E▪P▪AGE** paradigm, founded upon the concept of *sound science for an environmental engineering, effectual economics, and public policy agenda*, is used in testing and assessing the implementability of the engineering and economics tools via a real-world case study investigation.

1.3.1 TIME OF REMEDIATION FORMULATION

Using the Natural Attenuation Software (NAS) package developed at Virginia Tech, a set of formulae have been developed for calculating the TOR for petroleum-contaminated aquifers (specifically tracking benzene and MTBE) through statistical techniques. With the knowledge of source area residual saturation, groundwater velocity, and contaminant plume source length, the time to remediate a site contaminated with either benzene or MTBE can be determined across a broad range of regulatory maximum contaminant levels.

1.3.2 FRAMEWORK FOR DECISION ANALYSIS

After developing formulae for TOR, an integrated and interactive decision tool for framing the decision analysis component of the remediation problem was developed. While MNA can be a stand-alone groundwater remediation technology, significant benefits may be realized by layering a more traditional source zone remedial technique with MNA. Excavation and soil vapor extraction when applied to the front end of a remedial action plan can decrease the amount of time to remediation and while generally more expensive than an MNA-only approach, may accrue long-term economic advantages that would otherwise be foregone.

1.3.3 ASSESSMENT: SCIENCE ▫ ENGINEERING ▫ ECONOMICS ▫ POLICY

The value of these research components can be realized within the engineering and science communities, as well as through government, business and industry, and communities where groundwater contamination and remediation are of issue. The TOR formulation simplifies the

inputs necessary to determine the number of years that an MNA strategy will require before project closure and thus reduces the specialized skills and training required to perform a numerical analysis that for one set of conditions could require many hours of simulation time. The economic decision tool, that utilizes a life cycle model to evaluate a set of feasible alternatives, highlights the tradeoffs between time and economics can be realized over the lifetime of the remedial project.

CHAPTER TWO — LITERATURE REVIEW

2.1 THE EVOLUTION OF THE NATION'S ENVIRONMENTAL REGULATORY AGENDA

The history of the *environmental* movement in this country is well documented. Portney (1990) traces the regulatory beginnings of our federal government back to 1887 when Congress created the Interstate Commerce Commission, through the creation of the Federal Reserve Board and Federal Trade Commission, on through the "first great burst of federal regulatory activity" of the 1930s and finally the second major growth spurt of the 1970s and the creation of the United States Environmental Protection Agency.

Soesilo and Wilson (1997) tag hazardous waste site remediation as one of the most complex and controversial issues within the environmental arena. They identify "issues of health and safety, liability and equity, trust and fear, the reliability of remediation technology, and the need for urban economic development" as sensitive issues for society. They also provide a historical trace of environmentalism concurrent with the United States' economic uprising during the late 1800s, also referred to as the industrial revolution. The second World War provided the fuel for a rapidly growing economy during the first half of the 20th century, followed by unprecedented production of consumer goods, leading to huge quantities of municipal and industrial waste products. This serious threat to health and the environment led to passage of the Solid Waste Disposal Act (SWDA) in 1965.

In 1976, SWDA was amended by the Resource Conservation and Recovery Act (RCRA), which increased US EPA's purview of hazardous waste management (Soesilo and Wilson, 1997). With a major emphasis on prevention, RCRA attempts to minimize the risk to health and the environment by, first, preventing the generation of hazardous waste and, second, preventing the release of hazardous waste into the environment. The Comprehensive Environmental Response, Compensation and Liability Act (CERCLA or *Superfund*) became law in 1980 and was a response to the need to mitigate hazardous waste at abandoned sites. Superfund facilitated a taxing mechanism on the chemical industry and precipitated the cleanup of "hazardous waste sites by removing contaminated soils from the sites, and/or remediating the contaminated sites by using various treatment technologies."

In addition to RCRA and its 1984 Amendments, and CERCLA and its 1986 Amendments, two major federal laws, entrusted to the EPA, have been major drivers for the attainment of acceptable levels of water protection (NRC, 1999):

- Clean Water Act (1972) — Established requirements controlling discharge of pollutants to surface waters
- Safe Drinking Water Act (1974) — Developed drinking water standards that govern the quality of water delivered to the consumer and established the underground injection control program.

While neither directly regulates groundwater remediation, they do form the basis for setting groundwater and soil remediation goals through the establishment of public drinking water standards for contaminant compounds, referred to as *maximum contaminant levels* or MCLs. These MCLs are determined primarily by toxicity and carcinogenicity risk, drinking water treatment technology capacity, and considerations of cost.

2.2 The Costs of Groundwater Cleanup

Soesilo and Wilson (1997) illustrate the inherent difficulties – extremely lengthy and very costly – of hazardous waste site remediation via several examples that remain of historical significance. In the late 1970s, the nation was shocked by "the discovery of a large number of abandoned, leaking hazardous waste dumps – *toxic time-bombs waiting to go off* – that were contaminating the environment and threatening human health … scattered across the country" in places like New Jersey, Delaware, Michigan, Iowa, Tennessee, Louisiana, California, Oregon, and Washington (Brown, 1980).

Approximately 40 percent of a $2.6 billion US EPA budget (between 1987 and 1993) was used to build remedial solutions. Unfortunately, only 13 of the more than 200 sites received a portion of that $1.1 billion. Superfund has shifted much of the onus of cleaning up hazardous waste sites to the polluting parties. The median expenditure reported by Fortune 500 Industrial and Fortune 500 Service corporations involved in Superfund clean up has been $1.5 million, of which, $1 million was used for site clean up, while $500 thousand was used for legal expenses (US GAO, 1994).

Portney (1990) reported the cost of complying with federal air and water pollution-control regulations at $52 billion for the year, 1981, as reported in 1988 dollars and exclusive of the costs of pesticide, toxic substance, or hazardous waste regulations. US EPA (1984) estimated that almost 65 percent of $1 trillion dollars could be spent in one decade toward the pursuit of clean air and water. If our nation maintains its current course with respect to cleanup costs, it is projected that direct cleanup cost may reach $750 billion through the first two decades of this century (Soesilo and Wilson, 1997). Another estimate by EPA (1990b) projects "an investment of $115 billion a year in [1990 dollars] to protect and restore our nation's air, water, and land. Site characterization costs are included in this sum of actions that take place once the decision to remediate has been made. These costs, for any site, are a direct function of the remedial technology selected. Moreover, researchers at the University of North Dakota (Grasso, 1993) have reported that "compliance with government regulations represents the most important consideration in selecting site remediation technology, followed by economics, public acceptance, and the engineering aspect of the technology." Sarno (1991) makes the following statement about the selection of an appropriate remediation technology: "the decision maker must combine technical judgment, cost, liability, and statutory criteria to arrive at a decision."

The site remediation program has been and continues to be lengthy and costly as evidenced by "legal jurisdiction regarding the remediation process, uncertainty of science, how clean a remediated site must be before it is considered clean enough to be an acceptable level of risk, and liability and equity issues" (Soesilo and Wilson, 1997). Site remediations will neither cease nor begin to be free. Policy mandates will not evaporate, nor will social concerns become non-existent. The message to the engineering community is to build integrated teams of multi-disciplinarians to continue remedial actions while mitigating costs without compromising human health or the environment.

The groundwater and soil cleanup technology development program for the Department of Energy (DOE) is driven by "the need to meet applicable federal and state regulations for the cleanup of contaminated sites" (NRC, 1999). However, DOE has stated that it *cannot* meet all of the applicable regulations with the existing remedial technologies. Estimates as recent as 1999 have suggested that costs to cleanup sites owned by DOE, the Department of Defense (DoD), state governments, and private enterprises could well exceed $1 trillion dollars if these site managers were

required to remediate to the most stringent standards. While these estimates have been calculated using conventional and readily available remediation technologies, experienced experts have come to recognize that there are no available solutions to the most recalcitrant contaminants, irrespective of costs.

2.3 Two Types of Contaminants and Corresponding Remedial Approaches

2.3.1 NONAQUEOUS PHASE LIQUIDS

Subsurface contamination resulting from fuel hydrocarbons may exist in up to four different states (i.e., dissolved in water, sorbed onto the soil matrix, volatilized in soil vapors, or as nonaqueous phase liquids or NAPLS) (Wiedemeirer et al., 1999). NAPLs are a problem at many contamination sites because these organic liquids may not easily dissolve in water and may thus be present as an independent fluid phase (NRC, 1999). When NAPLs are introduced into the unsaturated zone they generally migrate down toward the saturated zone where they will either float on the water table, forming a thin, pancake-like layer that spreads across the water surface, or penetrate the water table and migrate into the saturated zone, to depths on the order of hundreds of feet (Wiedemeirer et al., 1999). These are called light and dense NAPLs, respectively.

NAPLs may be present in the subsurface as a single chemical; however, the majority of these contaminants are multiple-chemical mixtures. Gasoline is an example of a fuel hydrocarbon, almost all of which are mixtures, that is "composed of hundreds of different compounds with a wide variety of chemical and physical properties." Wiedemeirer et al. (1999) further summarizes the release of NAPLs into the subsurface:

- The "NAPL will initially form a large, connected body of continuous NAPL (also called free-phase NAPL, mobile NAP, nonresidual NAPL, or free product) that migrates through the subsurface, invading pore spaces, fractures, and particularly any preferential pathways in the subsurface matrix, and displacing water and air in the unsaturated zone and water alone in the saturated zone
- "If the source persists over a long period, or if a large volume of NAPL is released, the downward and lateral migration of the NAPL may be extensive, with light non-aqueous phase liquids (LNAPLs) – liquids which are sparingly soluble in water and less dense than

water (USGS, 2006a) – spreading across the water table and dense non-aqueous phase liquids (DNAPLs) – liquids that are denser than water and do not dissolve or mix easily with water (USGS, 2006b) – fingering downward into the saturated zone and spreading laterally across capillary barriers

- "Once the source loading ceases, however, the mass of continuous NAPL begins to stretch, and processes such as snap-off and bypass form residual NAPL in the subsurface zones through which mobile NAPL flowed."

Residual NAPL are an added source of complexity in site remediation. These "discrete blobs of NAPL in one or a few pores may act as a long-term source of soluble contaminants to groundwater because these individual blobs are not connected to each other and are rendered essentially immobile by capillary forces."

The early approach to groundwater remediation was to *pump-and-treat*. This technique gained credible acceptance, especially at Superfund sites, because the remediation community generally believed that most contaminants were released into the vadose zone, and because of precipitation, reached the saturated zone in either aqueous of sorbed phases, but not as NAPLs (Wiedemeirer et al., 1999). Mackay and Cherry (1989) were among the first researcher to delve into the constraints on aquifer remediation and the question of why pump-and-treat systems were not producing adequately restored aquifers. Through their research they highlight the following reasons for caution regarding the pump-and-treat option:

- At sites where a considerable amount of fugitive contaminant mass exists in the subsurface, the rate of mass removal by pumping will be significantly retarded.
- While pump-and-treat tends to reduce the size of the plume in the source area, the contaminant rebound and tailing effects necessitate that pumping be continuous thus creating a perpetual remedial strategy.
- If a pump-and-treat system is to operate in perpetuity it becomes financially prohibitive due to the continuous and massive volumes of water that must be extracted and treated in order to flush out the remaining contaminants.

Travis and Doty (1990) state emphatically that, relative to groundwater contamination, it is impossible to restore aquifers at Superfund sites to an environmentally sound condition utilizing

pump-and-treat. The US EPA would recognize this fact because pump-and-treat systems were so widely employed as a groundwater remediation technology, yet seeming unable to reduce contaminants to the required health-based limits within traditional design periods of approximately 5 to 10 years, decision makers would need a better foundation for evaluating the applicability of the pump-and-treat approach. Their document, "Pump-and-Treat Ground-Water Remediation: A Guide for Decision Makers and Practitioners," would illuminate many of the gaps from theory to the field (US EPA, 1996a). Travis and Doty (1990) provide a qualitative characterization of the phenomena at a typical pump-and-treat site – during pumping, concentrations would initially fall by a factor of 2 to 10 and subsequently level without any more decline unless pumping were to cease resulting in an increase in contaminant concentration. However the EPA guidance document (US EPA, 1996a) provide a more substantial characterization of the phenomena known as *tailing* – "the progressively slower rate of decline in dissolved contaminant concentration with continued operation of a pump-and-treat system" and *rebound* – "the fairly rapid increase in contaminant concentration that can occur after pumping has been discontinued … [but] may be followed by stabilization of the contaminant concentration at a somewhat lower level."

The US EPA guidance document (US EPA, 1996a) describes how tailing and rebound are influenced by the contaminant's physical and chemical characteristics, the subsurface solid matrix, and the properties of the groundwater. The factors and processes are illuminated with the aid of research and field experiences from Cohen et al. (1994), National Research Council (1994), Gorelick et al. (1993), Palmer and Fish (1992), Gillham et al. (1984), Keely (1989), Roy et al. (1992), Piwoni and Keely (1990), and Berglund and Cvetkovic (1995). The four major processes may be broadly classified by *contaminant desorption*, *precipitate dissolution*, *matrix diffusion*, *ground-water velocity variations*, or *the presence of NAPLs*:

- Contaminant Desorption: The movement of contaminants in the subsurface is retarded by sorption mechanics that can be reversed. "Thus as dissolved contaminant concentrations are reduced by pump-and-treat system operation, contaminants sorbed to subsurface media can desorb from the matrix into the ground water … causing the tailing of contaminant concentrations during pumping as well as rebound after pumping stops."
- Precipitate Dissolution: Precipitation-dissolution reactions are reversible in a similar manner to sorption-desorption mechanics; however, they are associated with inorganic compounds.

The tailing is controlled by solubility and can result in rebound if the solid phase is not depleted before the end of pumping.

- Matrix Diffusion: The fate of a contaminant through a porous media is highly dependent on the nature and degree of the heterogeneities of the solid matrix and most likely to be of influence with inorganic anions and some organic chemicals as the concentration gradients favor the diffusion of contaminants to the less permeable medium. "During a pump-and-treat operation, dissolved contaminant concentrations in the relatively permeable zones are reduced by advective flushing, causing a reversal in the initial concentration gradient and slow diffusion of contaminants from the low to high permeability media." The influence of matrix diffusion is directly proportional to the temporal span between contamination and remediation. "In heterogeneous aquifers, matrix diffusion contributions to tailing and rebound can be expected, as long as contaminants have been diffusing into less-permeable materials."
- Ground-Water Velocity Variation: Tailing and rebound are detected in aquifers as a result of differences in travel times that occur as a result of the multiple flow paths that can be taken by a contaminant under the influence of the pumping of an extraction well.
- Non-Aqueous Phase Liquids: NAPLS are considered immiscible and relatively insoluble in water. They are also sufficiently soluble such that groundwater concentrations often exceed regulatory limits. "Consequently, residual and pooled free-product NAPL will continue to contaminate ground water that makes sufficient contact to dissolve small amounts from the NAPL surface. When ground water is moving slowly, contaminant concentrations can approach the solubility limit for the NAPL. Although pump-and-treat systems increase ground-water velocity, causing an initial decrease in concentration, the decline in concentration will later tail off until the NAPL's rate of dissolution is in equilibrium with the velocity of the pumped ground water. If pumping stops, the ground-water velocity slows and concentrations can rebound, rapidly at first and then gradually reaching the equilibrium concentration, unless pumping is resumed."

US EPA would follow with an issue paper (Huling and Weaver, 1991) and workshop (US EPA, 1992) discussing the difficulties in detecting and removing DNAPLs in groundwater. Later, EPA would follow with an issue paper addressing site characterization and remediation of LNAPs at

sites contaminated by petroleum hydrocarbons (Newell et al., 1995). It is now widely accepted that NAPLs must be addressed at the majority of sites contaminated with petroleum hydrocarbons.

2.3.2 LIGHT NONAQUEOUS PHASE LIQUIDS

The most common LNAPLs at sites with groundwater contamination are gasoline and petroleum hydrocarbon mixtures (Charbeneau et al., 1992). The subsurface fate and transport mechanisms are influenced by the liquid properties (e.g., density, viscosity and interfacial tension), soil properties (e.g., porosity and permeability), liquid-soil properties (e.g., capillary pressure, relative permeability, wettability, saturation, and residual saturation), and site-specific physical properties (e.g., depth to groundwater, volume of release, and direction of groundwater flow) (US EPA, 1996b). Many LNAPL components are biodegradable, "primarily because they have been around the earth as natural substances for millions of years, and bacteria have developed the necessary methods to break them down and use them as an energy source" (Philips, 2002). Microbes are capable of biodegrading the individual components of the LNAPL mixture when bio-available. Benzene is a common component in gasoline known to biodegrade. It is known as the parent compound of the aromatic series and together with toluene, ethylbenzene, and the three isomers of xylene, comprise about 15 percent of gasoline (Testa and Jacobs, 2006). MTBE is an exception as it is very soluble in water and not easily biodegraded.

MTBE is a synthetic compound that was designed to address vehicle emissions and clean air standards. While indeed improving air quality, MTBE has become a major problem for ground and drinking water. MTBE is at least 75 times more water soluble than many of the other components of gasoline and has been found in groundwater at over 250,000 contaminated sites (Jacobs et al., 2001). Unlike, benzene which is a known carcinogen, the health effects data for MTBE has been inconclusive; however, taste and odor considerations have been the primary regulatory driver as health effects continue to be understood.

Wilson (2003) describes a number of mechanisms that attenuate the concentrations of MTBE and other gasoline constituents once dissolved in ground water. Gasoline constituents advance with the wetting front in the direction of flow. Dilution and dispersion attenuate the constituents along the flow path. Also, attenuation may occur through adsorption to aquifer solids

and through aerobic and anaerobic biodegradation. While the "distribution of MTBE and other gasoline constituents in groundwater is controlled by hydraulics and natural attenuation processes acting in groundwater, the persistence of contamination is usually controlled by the rate of transfer of constituents from the residual gasoline to the moving ground water." This persistence over time is generally due to regeneration from the residual gasoline that remains in the source area.

2.4 NATURAL ATTENUATION

Natural attenuation occurs to some degree at all contaminated sites (Odencrantz et al., 2002). It refers to the "reduction of contaminant concentration and contaminant mass by relying on the assimilative capacity of a groundwater system" (Wiedemeier et al., 1999). This is an in-situ process that occurs without human intervention, and reduces either the mass, toxicity, mobility, volume or concentration of a contaminant. It is considered relatively nonintrusive, very cost-effective with respect to traditional remediation technologies, and more protective of health and the environment since contamination is not transferred from one medium or phase to another. Other terms for natural attenuation include intrinsic remediation, intrinsic bioremediation, natural restoration, or passive bioremediation.

Monitored natural attenuation incorporates natural attenuation as a remedial approach to plume management that involves long-term plume monitoring. US EPA defined MNA in 1997 in the Office of Solid Waste and Emergency Response Directive, *Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites*:

> *The term monitored natural attenuation, as used in this Directive, refers to the reliance on natural attenuation processes (within the context of a carefully controlled and monitored site cleanup approach) to achieve site-specific remedial objectives within a time frame that is reasonable compared to that offered by other more active methods. The "natural attenuation processes" that are at work in such a remediation approach include a variety of physical, chemical, or biological processes that, under favorable conditions, act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil and groundwater. These in-situ processes include biodegradation; dispersion; dilution; sorption; volatilization; and chemical or biological stabilization, transformation, or destruction of contaminants.*

Within the past decade, natural attenuation has been gaining acceptance as a stand-alone remedy or as a component of combined remedial strategies (Surampalli et al., 2004). This is certainly

true for petroleum hydrocarbons and has been spurred by developments over the past three decades that include (1) increased national attention to *the groundwater contamination problem* including underground storage tanks that leaked petroleum hydrocarbons into the subsurface; (2) the concurrent growth in research pertaining to the potential of subsurface microorganisms to help remediate petroleum hydrocarbon-contaminated sites; and (3) a period of unprecedented assessment and monitoring of sites in the United States, Canada, and Europe contaminated by petroleum hydrocarbons that showed slower than expected plume migration, steady state convergence, areal contraction or a temporal decrease in contaminant concentration.

Moreover, several factors were important to the acceptance of MNA as an US EPA-approved treatment solution as reported by Surampalli et al. (2004):

- "There was a lack of efficient, cost-effective engineered remedial technologies for cleanup of the large number of contaminated sites."
- The seminal report, "In situ Bioremediation: When does it work?," (NRC, 1993) "clearly distinguished intrinsic bioremediation from engineered bioremediation as an approach that, 'manages the innate capabilities of naturally occurring microbial communities to degrade environmental pollutants without taking any engineered steps to enhance the process.'"
- In response to the large number of contaminated sites and the limited availability of funds to cleanup BTEX sites, risk-based corrective action (RBCA) programs were developed and implemented in order to categorize and prioritize leaking underground storage tank site cleanups.

Thus, natural attenuation has been used in conjunction with RBCA to predict both the migration of subsurface petroleum hydrocarbons and provide an estimate of the possible impact on affected receptors.

The biodegradation of petroleum hydrocarbons is a key natural attenuation process represented by the following general equation (Wiedemeier et al., 1999):

$$\text{microorganism} + \text{electron donor} + \text{electron acceptor} + \text{nutrients} \rightarrow$$
$$\text{metabolic by-product} + \text{energy} + \text{microorganisms}.$$

However, all of the active processes contribute to a time of remediation (TOR) or a time at which the NAPL contaminant has dissolved, dispersed and/or biodegraded below a regulatory-

determined level at an associated receptor location (Chapelle et al., 2003). The abiotic processes, such as advection, dispersion, sorption, dilution, and volatilization, must also be factored into reliable estimates of TOR.

2.5 TIME OF REMEDIATION

The time required to achieve acceptable levels of contaminant occurrence or exposure relate directly to public health and ecological goals. Furthermore, this TOR is essential to determining the economic costs of site cleanup. Empirical and deterministic techniques have been developed to assess TOR at sites with depleting sources. Empirical approaches involve regression analysis of long-term trends in contaminant data at source monitoring wells to assess source strength over time and estimate TOR in relation to site-specific remediation objectives (US EPA 2002a). For example, Wilson and Kolhatkar (2002) report that source attenuation rates based on MTBE concentration data at five UST sites ranged from 0.15 to 0.75 yr^{-1}, assuming that source depletion follows a first-order decay function. Difficulties with the empirical approach include climatic-induced water level fluctuations and the subsequent impact on observed source zone contaminant concentrations and the inability to demonstrate statistically-significant rates due to a lack of sampling dates or high source strength (US EPA 2005b). In addition, it has not been established that a first-order decay model is appropriate for source depletion over remediation timeframes observed at NAPL sites.

Deterministic approaches to estimating TOR for source attenuation are based on mass balance equations for the source zone and may include both analytical and numerical solutions. Brouwers and Augustijn (2001) present an analytical solution for a single-component NAPL coupled to the one-dimensional solute transport equation to quantify when a NAPL source zone is flushed with clean groundwater. Chapelle et al. (2003) describe an interactive computer program (NAS, Natural Attenuation Software) designed to implement a numerical code (SEAM3D, Sequential Electron Acceptor Model in 3D) for simulating the TOR problem for a multi-component NAPL source zone. By applying NAS to a well-characterized gasoline-contaminated site in South Carolina (Laurel Bay), the authors demonstrate that the most soluble components (in this case, MTBE and benzene) with relatively-low mass fractions deplete rapidly from the source zone when compared to less soluble components in the gasoline (i.e., toluene, ethylbenzene, and xylene). This is consistent with the observations described in Landmeyer et al. (1999) that show the depletion of MTBE and benzene from the source zone during a long-term monitoring program at the Laurel Bay site.

Mendez et al. (2004) present the application of NAS at the Laurel Bay site to simulate the depletion of benzene concentrations at a source-area monitoring well based on estimates of groundwater velocity and LNAPL mass, composition, and dimensions.

2.6 Economic Analysis

If a robust economic analysis can shed insight into *whether an action should be taken to achieve an intended objective*, or *which action should be taken to achieve an intended objective*, or once an action has been taken, *how much of the action and for how long should the action be applied to achieve the intended objective?* then it is considered a cost effective analysis.

The cost-benefit analysis is an effective tool for use when the economic analysis is at the strategic level as the concern is whether the benefits of achieving the objective are worth the costs of the actions required to meet such a goal (Hardisty and Özdemiroğlu, 2005). Alternatively, cost-effectiveness or least-cost analysis is generally applied, tactically, by simply selecting the most economical option that will achieve the mandate.

Where costs are applied to a remediation project, time horizon is an important consideration. Many remedial projects project out to 30 years. This is often beyond the traditional budget considerations of most parties financially responsible for contaminant site cleanup. While traditional remediation projects have a large initial investment (or capital) cost that is due toward the beginning of a remedial action project, the operation and maintenance (O&M) costs will be spread out over the life of the project. To compare these site- and technology-variable costs, it is important to use discounting, and generally in a net present value framework.

Discounting allows present and future monetary values to be compared using a mechanism that places greater weight on present amounts relative to future amounts. The discounting principle is motivated by two reasons (Hardisty and Özdemiroğlu, 2005):

1. The productivity of capital argument states that "one unit of currency that is invested today will be worth more next year due to the interest on capital"
2. The time preference argument "relies on the observation of the behavior of individuals and concludes that, regardless of interest rates, people do prefer their benefits now rather than later, simply because people have limited lifetimes and hence are impatient."

Remediation engineers may be aided by the *Life Cycle* concept which was first published, in reference to groundwater, in *Groundwater Treatment Technology* (Nyer, 1985). The basis for the Life Cycle concept follows (Nyer, 1997):

- Groundwater contaminant cleanups are unique
- Project requirements do change over the life of the project
- Project design must factor in project life, not only initial project conditions.

The natural progression is that this Life Cycle analysis may provide a highly-favored estimate of when to cease active remediation and is illustrated below.

The Federal Remediation Technologies Roundtable (FRTR, 2005a) has been working "to build a collaborative atmosphere among federal agencies involved in hazardous waste site cleanup … [which] have led to technology development and demonstration partnerships with private developers, a more consistent and unified federal approach to technology evaluation and regulatory acceptance, and a variety of technology transfer tools and other information resources." US EPA currently provides access to environmental online software and tools such as Scribe: Environmental Field Data Capture, Natural Attenuation Software, Monitoring and Remediation Optimization System, Rapid Assessment Tools, Mass Flux Toolkit, Sustainable Management Approached and Revitalization Tool – electronic, Field Operations and Records Management Systems II Lite, Hazard Ranking System Quickscore, Spatial Analysis and Decision Assistance, Pro Upper Confidence Level Software, Environmental Knowledge and Assessment Tool, DOE Visual Sample Plan Software, Fully Integrated Environmental Location Decision Support, and The OnSite On-line Tools for Site Assessment (TIP, 2008).

Among the tools for estimating environmental costs are Environmental Cost Element Structure, Remedial Action Cost Engineering Requirements (RACER) System, Cost Risk, Micro-Computer Aided Cost Estimating System, Cost-To-Complete System, and IDEAL Estimating System (FRTR, 2005b). Additionally, the FRTR website provides guidelines and references for a number of Technology Cost Analyses (FRTR, 2007). These include links to the Army Cost and Economic Analysis Center, Assistant Secretary of the Air Force (Financial Management and Comptroller), Compendium of Cost Data for Environmental Remediation Technologies, Hazardous, Toxic, and Radiological Waste Work Breakdown Structure, Naval Center for Cost

Analysis, Tri Services Automated Cost Engineering Systems, US Army Corps of Engineers Area Cost Factors, and US Department of Energy/Environmental Management (DOE/EM) Applied Cost Engineering Team. DOE/EM has facilitated the collaborative efforts of program and project managers, cost engineers, and cost estimators to develop credible cost estimates and subsequently manage the associated projects and programs (ACE, 2008). RACER is the most relevant of online/software tools for this research. According to the developers of RACER, Earth Tech, Inc. (2004), "the software is a Windows-based environmental remediation/corrective action cost estimating system [that] quickly and accurately estimates costs for all phases of environmental remediation projects – from site investigation through site closeout. The system allows users to develop and update cost estimates, evaluate and compare the cost of various treatment options, quantify environmental liability for budgeting or regulatory/financial disclosures, and develop a consistent approach for project budgeting." However, it still lacks the total integration desired (DRC, 1995) and "is not meant for estimating costs of the entire life-cycle."

The online tools that could be of benefit with regard to life-cycle costing and assessment are more strongly engineered for the construction industry. The website, "Whole Building Design Guide: The Gateway to Up-To-Date Information on Integrated *Whole Building* Design Techniques and Technologies,"(NIBS, 2008) provides access to several programs that incorporate life cycle economic techniques, such as Building for Environmental and Economic Sustainability, Building Life-Cycle Cost, and Life Cycle Cost in Design. However, the applicability to environmental remediation projects is still lacking.

CHAPTER THREE — TIME OF REMEDIATION FORMULATION

3.1 A STATISTICS BASED FORMULAE FOR CALCULATING TIME OF REMEDIATION

The first research objective helps to define the major theme of this research. Through computer simulations and statistical techniques, the aim of this research is to develop formulae for calculating the time of remediation (TOR) for a petroleum-contaminated aquifer, specifically tracking benzene and MTBE. The TOR, as an estimate of the time required to remediate a site to regulatory acceptable limits, is incredibly important in the design and execution of a site remediation strategy. This Phase of the research is a deterministic approach to develop a regression-based solution to the TOR problem that is simpler to use by engineers and scientists, but especially non-technical affected and interested stakeholders.

The field of *remediation engineering* (Suthersan, 1997) has grown and developed into a discipline that has advanced technology applications and created a wealth of information that chronicle the successes and failures of past, current, and evolving technologies. As the number of recognized waste sites continue to grow so do the influences that drive regulatory considerations, social concerns, feasibility constraints, and other factors (Wise et al., 2000). Regardless of the difficulties of engineering remediation solutions to the dynamic and complex waste sites of the twenty-first century, *protection of human health and the environment* is still the most critical of drivers that contribute to effective environmental policies and remediation processes (US EPA, 2008).

The use of integrated remediation techniques is critical to the success of present and future efforts to meet compliance requirements and preserve the integrity of our environment for human and non-human utilization (Avogadro and Ragaini, 1993). In many situations even the most innovative of technologies are unaffordable for remediating many of the nation's waste sites (Yong and Mulligan, 2004). The emergence and acceptance of natural attenuation processes, as a legitimate mechanism for cleaning up contaminated soils and groundwater, point toward a new era of environmental cooperation among policy makers, regulators, and responsible and affected parties. Consequently, it is imperative that better and more efficient mechanisms for characterizing the critical aspects of a monitored natural attenuation (MNA) strategy be further investigated.

Time of remediation must be determined to implement an MNA approach to groundwater cleanup. While models (conceptual, numerical, etc.) are a present-day necessity for environmental planning and management, the most complex models require the most resources (time, training, data requirements, costs) to employ appropriately. It is imperative that a more efficient mechanism for determining the TOR of a contaminant zone be investigated. Through the use of simple statistical models, it is proposed that TOR can be accurately estimated from and correlated to the results of the Natural Attenuation Software (NAS) developed at Virginia Tech (Widdowson et al., 2008). Additional information on NAS is provided later in Section 3.2.1. NAS is a screening tool designed to estimate TOR by employing a deterministic approach to solute transport and attenuation combined with depletion of a multi-component light non-aqueous Phase liquid (LNAPL) source.

Through the developed formulae, it will be possible to significantly reduce the time, costs, and minutiae that are required to run a complex computer program such as NAS and present a simplified mechanism for a wider range of interested parties to utilize. It is expected that the results of this statistics based paradigm will be capable of providing a reasonable and highly accurate estimate of the TOR for a NAPL, specifically, benzene and MTBE.

The conceptual model for the TOR problem is illustrated in Figure 3-1. Using site-specific hydrogeologic data, contaminant source information, contaminant geochemistry, and a given regulatory compliance concentration (RCC), *the time required for contaminants in the source area to attenuate to a predetermined target source concentration* (Widdowson et al., 2007) can be determined by solving the advection-dispersion equation coupled to a conversation of mass equation for each component in the NAPL source. By estimating and furthermore understanding this TOR, it is possible to evaluate the efficacy of contaminant treatment options with respect to their adverse impact on human health and the environment down gradient of the contaminant source.

Especially evident in the aforementioned figure is the conceptual construct that relates NAPL mass reduction to NAPL concentration reduction via an associated TOR. As mass is reduced in the contaminant source area, an associated reduction in the groundwater concentration of the contaminant is also observed. The time at which this concentration reaches or falls below the

RCC is then the TOR that provides to responsible, interested and affected the parties the timeframe that will be required to achieve the corresponding remedial action objectives.

Using a numerical model for multi-component (such as benzene and MTBE) solute transport and NAPL dissolution, a generic source zone model was constructed. Simulations of concentration versus time were performed for a wide range of field-scale parameters. Equations for estimating the time to reach microgram per liter concentrations of either MTBE or benzene in the source zone are derived from a multi-step regression analysis of the computationally-derived results.

3.2 A METHODICAL APPROACH TOWARD A TIME OF REMEDIATION FORMULAE

Engineering data are often composed of a systematic and random component (McCuen and Snyder, 1986). Our interest is in the degree to which multiple-parameter TOR data generated through NAS might be explained through three independent variables: residual saturation, contaminant source length, and groundwater velocity. Multiple simulations were conducted via NAS, and TOR data was generated for a given set of parameters and variables as defined in Tables 3-1 and 3-2, respectively. The relationship between TOR and each of the aforementioned three independent variables was explored. Accordingly, the most appropriate independent – dependent relationship was plotted and statistically analyzed. Regression coefficients were determined and iteratively reassessed graphically and statistically for quantitative relationships that exist with respect to either length or velocity. This analytical procedure was repeated for the last of the three independent variables. Once a defensible equation for TOR with respect to residual saturation, contaminant source length, and groundwater velocity was derived, it was then used to estimate an empirical TOR and subsequently compared to the numerically-generated TOR from NAS. Finally, multiple regulatory standards or regulatory compliance concentrations (RCCs) were tested for the emergence of a repeated pattern of quantitative and qualitative significance.

3.2.1 OVERVIEW OF NAS

Natural Attenuation Software (NAS)—*an interactive computer program that estimates times of remediation associated with petroleum hydrocarbon and chlorinated ethane-contaminated aquifers*—was developed at the Virginia Polytechnic Institute and State University (Virginia Tech) in collaboration with the United States Geological Survey (USGS) and is used to estimate the time of remediation for

benzene- and MTBE-contaminated source zones to a regulatory-permissible level (Chapelle et al., 2003; Widdowson et al., 2007). It was originally conceived as a screening tool. A detailed set of inputs relative to hydrogeology, redox conditions, and contaminant characterization are needed for accurate and reliable results pertaining to the *timeframes required for natural attenuation processes, such as dispersion, sorption, and biodegradation, to lower contaminant concentrations and mass to predetermined regulatory goals in groundwater systems* (Chapelle et al., 2003). NAS is a tool designed primarily for remedial project managers and their site contractors and offers an effective mechanism for communicating the complex analytical and numerical solutions to the TOR problem (Widdowson et al., 2007).

3.2.2 EQUATIONS OF SOLUTION

The mathematical model that describes the NAS formulation is based up on the following:

$$D_x \frac{\partial^2 C_i}{\partial x^2} + D_y \frac{\partial^2 C_i}{\partial y^2} - v_x \frac{\partial C_i}{\partial x} - \lambda_i C_i + R_{source,i}^{NAPL} + k^{NAPL}\left(C_i^{eq} - C_i\right) = R_i \frac{\partial C_i}{\partial t} \qquad [3\text{-}1]$$

where C_i is the aqueous Phase hydrocarbon concentration [M/L^3] for ls = 1,2,...,NS (number of hydrocarbon compounds); x is distance in the longitudinal direction [L]; y is distance in the transverse direction [L]; t is time [T]; D_x and D_y are the longitudinal and transverse hydrodynamic dispersion coefficients [L^2/T], respectively; v_x is the groundwater velocity [L/T]; λ_i is the first-order decay rate [T^{-1}]; $R_{source,i}^{NAPL}$ is a hydrocarbon source term due to NAPL dissolution [M L^{-3} T^{-1}]; and R_i is the retardation factor [-].

For each hydrocarbon compound the driving force for dissolution is the difference between the equilibrium concentration (C_i^{eq}) and the aqueous Phase concentration (Imhoff et al. 1993; Parker et al. 1991). In general, the rate of dissolution of individual NAPL components into groundwater depends on the interfacial area between the NAPL and water, (Imhoff et al. 1993), aquifer heterogeneity (Mayer and Miller 1996), the size and shape of the NAPL blobs (Powers et al. 1994) and the groundwater velocity (Pfannkuch 1984). Thus, if transport processes occur at a high rate relative to the NAPL dissolution rate, C_i may remain lower than C_i^{eq}. This effect can be described mathematically so that the NAPL dissolution term in Equation 3-1 is expressed as

$$R_{source,i}^{NAPL} = max\left[0, k^{NAPL}\left(C_i^{eq} - C_i\right)\right] \quad [3\text{-}2]$$

where k^{NAPL} the mass transfer rate coefficient [T^{-1}]. Using Raoult's Law, C_i^{eq} is calculated (Corapcioglu and Baehr 1987; Parker et al. 1991) as

$$C_i^{eq} = f_i C_i^{sol} \quad [3\text{-}3]$$

where f_i is the mole fraction of hydrocarbon compound *i* in the NAPL [mol mol_{NAPL}^{-1}]; and C_i^{sol} is the solubility of hydrocarbon compound *i* in pure water. During each time step, f_i is computed as

$$f_i = \frac{C_i^{NAPL} / \omega_i}{I^{NAPL} / \omega_I + \sum_{ls=1}^{NS} C_{lc}^{NAPL} / \omega_i} \quad [3\text{-}4]$$

where C_i^{NAPL} is the NAPL mass of hydrocarbon compound *i* per unit mass dry soil [M M_{solid}^{-1}]; I^{NAPL} is the NAPL concentration of inert (i.e., relatively insoluble constituents) [M M_{solid}^{-1}]; and ω_j is the molecular weight of NAPL constituent *j*. Equations 3-3 and 3-4 represent the concept that the effective solubility of any NAPL constituent is reduced when other constituents are simultaneously dissolving into the aqueous Phase. With each time step, as mass depletes from the NAPL and is transferred to the aqueous Phase, the mass balance equation for the NAPL Phase is solved and the NAPL mass concentration variables (C_i^{NAPL}) are updated using

$$\frac{dC_i^{NAPL}}{dt} = -\frac{\theta}{\rho_b} R_{source,i}^{NAPL} \quad [3\text{-}5]$$

where ρ_b is the bulk density of the porous medium [M_{solid} L^{-3}]; and θ is aquifer porosity [L^o].

3.2.3 NAS – PARAMETERS AND VARIABLES

The parameters required for NAS simulations are listed in Table 3-1. Constant values of the following parameters were used: aquifer thickness, decay rate, dispersivity ratio, effective porosity, fraction of organic carbon, hydraulic gradient, longitudinal dispersivity, NAPL source width, single zone natural attenuation capacity, and porosity. These values remain constant throughout all the trials for both benzene and MTBE. The justification for these parameters being selected to remain

constant is based on conditions typical of contaminated field sites and NAS results that show little variability with respect to parameter range changes.

The saturated thickness of an aquifer can vary from less than one to hundreds of meters. One of the largest aquifers in the United States, the Ogallala, is one such formation that varies from less than 1 meter to approximately 160 meters (NPGCD, 2008). A porosity value of 25 percent captures an extensive range of subsurface materials that can be modeled. This value is at the lower range of unconsolidated sand and gravel deposits and toward the upper range of rocks such as fractured basalt, karst limestone, and sandstone (Freeze and Cherry, 1979). The effective porosity is taken to be the same as the porosity which is consistent with discussions by Fetter (1988) that suggest that in many cases there is not a significant decrease in water flow through the soil matrix relative to non-interconnected or dead-end pores.

The decay rates for benzene and MTBE may be similar or variant. Chen et al. (2005) show benzene decay rates of 0.0087 and 0.0339 day^{-1} and MTBE decay rates of 0.0021 and 0.0048 day^{-1}. While there is some spread in those numbers, a conservative approach is taken with both as benzene tends to be the risk driver and MTBE to capture the lowest rate in the published range. Freeze and Cherry (1979) report typical longitudinal dispersivity values in the range of 0.0001 to 0.01 meters and a dispersivity ratio between 5 and 20. However, they also report that field dispersivities are often much higher than laboratory ranges and can be as large as 100 meters. Thus, a middle value was selected.

The case specific variables that are inputs to the NAS model are given by Table 3-2. Each trial run may be defined according to the NAPL source length (L), groundwater velocity (v_xV), and residual saturation (Sr) for a specific regulatory compliance concentration (RCC). The ranges selected were based on average (or typical) hydraulic values for a wide range of hydrogeologic conditions as determined at field sites (Borden and Kao, 19922; Fetter, 1988; Freeze and Cherry, 1979; Tie and Morrow, 2005) . There are four regulatory levels for both benzene and MTBE.

The base cases for benzene and MTBE use an RCC of 5 μg/L and 20 μg/L, respectively. The RCC for benzene is an enforceable standard based upon US EPA's maximum contaminant level (MCL) as established by the Safe Drinking Water Act (OGWDW, 2006). The remaining values

for RCC were determined based upon a spread of applicability established in part by the action levels in many states that sought to reach to an MCL of 1 μg/L (AEHS, 2008). Thus values that were approximately, half, twice and an order of magnitude greater than the base case were selected.

The base RCC for MTBE is based upon a drinking water advisory MCL for odor threshold (OUST, 2008). The minimum value was selected because of an effort by states, such as California and New Hampshire to implement mandatory health-based MCLs at 13 μg/L (DDNREC, 2000). The use of 40 μg/L was based upon the Drinking Water Advisory threshold for taste (OUST, 2008). The maximum value was selected for five times the base case for a wider analytical range.

Nine residual saturation values were explored, along with six values for source length and six values for groundwater velocity. These ranges were selected due to their *average* characteristics that are typical for multiple remediation environments.

3.2.4 NAS – TOR MODEL RESULTS

The TORs, in years to cleanup, for a benzene RCC of 5 μg/L are provided in Table 3-3. Similarly, the TORs for an MTBE RCC of 13 μg/L are shown in Table 3-4. Along with the constant parameters from Table 3-1 and the regression variables from Table 3-2, these RCCs constitute the base cases for each contaminant. Tables 3-5, 3-6, and 3-7 show the TOR results for benzene RCCs of 2, 10, and 50 μg/L, respectively. Tables 3-8, 3-9, and 3-10 list the TOR results for MTBE RCCs of 13, 40, and 100 μg/L, respectively.

As illustrated by Figure 3-2, the relationship between TOR and residual saturation is highly linear and well-correlated. Also, the relationship between TOR and length is highly linear and well-correlated. However, the relationship between TOR and velocity is non-linear, although it is well-correlated, such that

$$TOR_{Benzene_{MCL}} = f(Sr, L, V) \quad [3\text{-}6]$$

Similarly, data for MTBE, as verified through Figure 3-3, suggest that the relationships established for benzene may be similarly applied, such that

$$TOR_{MTBE_{MCL}} = f(Sr, L, V) \quad [3\text{-}7]$$

3.2.5 DEVELOPMENT OF TOR REGRESSION STEPWISE EQUATIONS

In the first phase of the regression analysis, it is evident that the independent-dependent relationships are evident across the entire range of variables. As shown in Figure 3-4, it is evident that the plots of time of remediation versus residual saturation for each velocity and a corresponding length are well-conditioned for definite quantitative model. Because of the strong correlation of TOR to Sr, it is the first step in defining the quantitative relationship, such that

$$TOR_{Benzene_{MCL}} = f(Sr) \quad [3\text{-}8]$$

Moreover, because this relationship is linear, the above generalization may be specifically defined according to

$$TOR_{Benzene_{MCL}} = a_B * Sr + b_B \quad [3\text{-}9]$$

The variable *a* is the slope of the linear regression line through the data points, and *b*, the intercept is the point at which the line of best fit will intersect the y-axis. They are shown in the following formulas (Downing and Clark, 1997; DeGroot and Schervish, 2002):

$$a = \frac{\sum (x-\bar{x})(y-\bar{y})}{\sum (x-\bar{x})^2} \quad [3\text{-}10]$$

$$b = \bar{y} - a\bar{x} \quad [3\text{-}11]$$

The line of best fit is assessed by the coefficient of determination, r^2, which reveals how closely the estimated value for the line of best fit corresponds to the actual data (DeGroot and Schervish, 2002):

$$r = \frac{\sum (x-\bar{x})(y-\bar{y})}{\sqrt{\sum (x-\bar{x})^2 \sum (y-\bar{y})^2}} \quad [3\text{-}12]$$

The relationship that is illustrated via Figure 3-4 for benzene is also valid for MTBE as shown through Figure 3-5 and given by

$$TOR_{MTBE_{MCL}} = f(Sr) \quad [3\text{-}13]$$

Moreover, because this relationship is linear, the above generalization may be specifically defined according to

$$TOR_{MTBE_{MCL}} = a_M * Sr + b_M \quad [3\text{-}14]$$

The second step of the analysis seeks to investigate the relationship between the slope and the intercept variables as a function of groundwater velocity, V:

$$a_L = f(V) \quad [3\text{-}15]$$

$$b_L = f(V) \quad [3\text{-}16]$$

The following regression relationships are applicable to both benzene and MTBE as a general construct was being sought by which to describe the relationship of the previously determined slope and intercept variables with respect to groundwater velocity, independent of contaminant. The more prominent relationships explored with respect to slope and intercept were (1) the linear function:

$$\begin{aligned} a_L &= \alpha_1 * V + \alpha_2 \\ b_L &= \beta_1 * V + \beta_2 \end{aligned} \quad [3\text{-}17]$$

(2) the power function

$$\begin{aligned} a_L &= \alpha_1 * V^{\alpha_2} \\ b_L &= \beta_1 * V^{\beta_2} \end{aligned} \quad [3\text{-}18]$$

(3) the logarithm function

$$\begin{aligned} a_L &= \alpha_1 * \ln(V) + \alpha_2 \\ b_L &= \beta_1 * \ln(V) + \beta_2 \end{aligned} \quad [3\text{-}19]$$

and (4) the exponential function

$$\begin{aligned} a_L &= \alpha_1 * e^{\alpha_2 \cdot V} \\ b_L &= \beta_1 * e^{\beta_2 \cdot V} \end{aligned} \quad [3\text{-}20]$$

The relationships are illustrated in Figure 3-6 showing the results of this initial Phase 2 analysis of the slope for a given length, a_L, regressed against velocity values linearly and non-linearly. Also shown is the intercept for a given length, b_L, regressed against velocity values linearly and non-linearly. With each successive phase progression some degree of correlation diminishes. While the linear regression from Equation 3-17 may not be the most robust characterization of the correlation

it does allow for a simplified mechanism to explore the formulation during these initial phase analyses.

The initial Phase 3 analysis explores the relationship between the Phase 2 coefficients, α_1, α_2, β_1, and β_2 and plume length, *L*. Figure 3-7: illustrates the correlation of the Phase 2 coefficients taken from Equation 3-17 and length. The linear function shows a strong correlation of the Phase 2 variables and source length, such that

$$\alpha_1 = f(L) = A_{11} * L + A_{12} \quad [3\text{-}21]$$

$$\alpha_2 = f(L) = A_{21} * L + A_{22} \quad [3\text{-}22]$$

$$\beta_1 = f(L) = B_{11} * L + B_{12} \quad [3\text{-}23]$$

$$\beta_2 = f(L) = B_{21} * L + B_{22} \quad [3\text{-}24]$$

Although more correlative power is lost with this last step, the use of the linear estimators does provide value added. The preliminary results that are illustrated in Figure 3-8 show that this backward stepwise regression model can lead to reasonable approximations of TOR when compared to the NAS-generated TOR estimates. However, the possibility of a more robust formulation leads to the exploration of an alternative method for conducting the Phase 2 analysis.

In performing this step-build regression solution, several observations were made with respect to the Phase 2 and 3 analyses. First, while the correlation between the Phase 1 coefficients and velocity were approximate, there was a sufficient loss of accuracy as to be of concern. Second, due to the loss of correlative agreement in the Phase 2 analysis, the relatively strong linear correlation in Phase 3 would be less efficacious in combination with each of the previous step-build regression stages. Third, the combination of length and velocity can be optimized into a variable called, *travel time*, *t*. Fourth, if the Phase 1 coefficients are strongly and linearly dependent upon residual saturation, the Phase 2 coefficients strongly dependent on velocity, and the Phase 3 coefficients strongly and linearly dependent on length, perhaps there is a travel time-like variable, T, that can consolidate Phases 2 and 3, and simplify the final portion of the step-build regressive formulaic construction.

Consequently, the Phase 2 and Phase 3 analyses were merged and it was deemed plausible that a single variable incorporating both source length and groundwater velocity could be

determined and shown sufficiently highly correlated to the Phase 1 coefficients as to produce a effective formulation for calculating time of remediation:

$$a = f(T) \quad [3\text{-}25]$$

$$b = f(T) \quad [3\text{-}26]$$

Initial tests evaluated the slope and intercept coefficients from the aforementioned linearly correlated estimates against source length, groundwater velocity, and multiple combinations of each. The most effective regressor is a dimensionless time variable, T_D, that is derived from a form of the Brouwers and Augustijn (2001) modified Damkohler number (ω^*) which includes the corresponding Peclet number:

$$T_D = (1 + \tfrac{1}{\omega^*}) \quad [3\text{-}27]$$

where

$$\omega^* = -\tfrac{L}{2\alpha} * \left(1 - \sqrt{1 + \tfrac{4k\alpha}{V}}\right) \quad [3\text{-}28]$$

In the above formulation, a is the longitudinal dispersivity and k is the decay rate. Table 3-11 shows the dimensionless and dimensioned time values for each source length, groundwater velocity pair. Also included in the table are the corresponding Phase 1 slope and intercept values for benzene and MTBE. Both correlations are logarithmic and highly correlated. The data are presented graphically in Figures 3-9 and 3-10 for benzene and MTBE, respectively. Consequently, the development of a formulae for calculating TOR for benzene and MTBE is based up on the relationships established during Phase 1 utilizing Equations 3-4 and 3-9, respectively, along with the Phase 2 results:

$$a_{Benzene} = 1532.5 * \ln(T) - 9573.9 \quad [3\text{-}29]$$

$$b_{Benzene} = 2.8793 * \ln(T) - 18.024 \quad [3\text{-}30]$$

$$a_{MTBE} = 76.882 * \ln(T) - 481.03 \quad [3\text{-}31]$$

$$b_{MTBE} = 4.0445 * \ln(T) - 25.284 \quad [3\text{-}32]$$

For more general applicability, the data for RCC values other than 5 μg/L (benzene) or 20 μg/L (MTBE) were used to determine time of remediation, not only for a given combination of residual saturation, groundwater velocity, and NAPL source length, but also for variable RCCs. The time of remediation ratios were calculated for benzene RCCs of 2, 10, and 50 μg/L with respect to the benzene-5 RCC. Likewise, the time of remediation ratios were calculated for MTBE RCCs of 13, 40, and 100 μg/L relative to the MTBE-13 RCC.

3.2.6 DEVELOPMENT OF TOR REGRESSION COMBINED EQUATIONS

Once it became clear that a strong correlative relationship does exist and can be used to estimate TOR for benzene and MTBE, the statistical analysis package, SAS (SAS, 2008), was used to evaluate the full set of data points for all values of Sr, T, and RCC. The above formulation is reduced to a singular step of statistical analysis.

Within SAS, a multiple regression, least squares model was built to evaluate all TOR data points (dependent) in relation to all other data points (independent). The most ideal relationship included Sr, ln(T), and RCC. Therefore, a full multiple regression analysis was performed on TOR using the *PROC REG* code (SAS, 2008) for the following regressors:

- Single: Sr ln(T) RCC
- Double: $Sr \cdot \ln(T)$ $Sr \cdot RCC$ $\ln(T) \cdot RCC$
- Triple: $Sr \cdot \ln(T) \cdot RCC$.

The R^2 value is the primary measure of the goodness of fit of the formulation.

3.3 TIME OF REMEDIATION FORMULATION RESULTS AND DISCUSSION

3.3.1 TOR REGRESSION – STEPWISE EQUATIONS

The results of the multi-stage backward step regression yield the following equation for estimating the time of remediation for benzene at a clean concentration of 5 μg/l (μg/L):

$$TOR_{Benzene_5} = 1532.5 \cdot \ln(T) \cdot Sr + 2.8793 \cdot \ln(T) - 9573.9 \cdot Sr - 18.024 \qquad [3\text{-}33]$$

where T = dimensionless travel time, and

Sr = residual saturation.

The dimensionless travel time variable is a function of source length, groundwater velocity, longitudinal dispersivity, and decay rate.

This equation may be modified to estimate the time of remediation for various regulatory RCCs. From the base 5 μg/L equation, estimates of time of remediation for benzene concentrations of 2, 10 and 50 μg/L were found to increase by 4.9%, and decrease by 4.5% and 15.3%, respectively. Table 3-12 lists the standard deviation for the conversion percentage. It is therefore possible to estimate the time of remediation of benzene given a clean concentration, contaminant source characteristics, and groundwater velocity. Furthermore, it has been determined that when these percentages are used to estimate the time of remediation for clean concentrations of 2, 10 and 50 μg/L, the average error is 0.8%, 0.03%, and 0.1%, respectively.

Similarly, the results of the multi-stage backward step regression yield the following equation for estimating the time of remediation for MTBE at a clean concentration of 20 μg/L:

$$TOR_{MTBE_{20}} = 76.882 \cdot \ln(T) \cdot Sr + 4.0445 \cdot \ln(T) - 481.03 \cdot Sr - 25.284 \qquad [3\text{-}34]$$

Estimates of time of remediation for MTBE concentrations of 13, 40, and 100 μg/L were derived from a 2.2% increase, and 3.5% and 8.3% decrease to the base 20 μg/L equation, respectively as stated in Table 3-13. Thus, it is possible to estimate the time of remediation of MTBE for a given a clean concentration, contaminant source characteristics, and groundwater velocity, as well. The average error using this methodology is 0.1% for all three conversions.

The results illustrated in Figure 3-11 indicate a strong agreement between the estimated values of time of remediation calculated for a benzene clean concentration of 5, 2, 10 and 50 μg/L and the time of remediation as determined by the Natural Attenuation Software. Figure 3-12 shows the presence of non-constant variance for the benzene data set. A similar set of plots is shown in Figures 3-13 and 3-14, illustrating the agreement between the estimated time of remediation for an

MTBE clean concentrations of 20, 13, 40, and 100 μg/L and the true value as determined by NAS, and the presence of non-constant variance.

3.3.2 TOR REGRESSION – COMBINED EQUATIONS

The multi-stage backward step regression yields the following equation for estimating the time of remediation for benzene:

$$TOR_{Benzene} = -6.052 \cdot Sr \cdot \ln(T) \cdot RCC + 1576.571 \cdot Sr \cdot \ln(T) + 37.841 \cdot Sr \cdot RCC - 0.006 \cdot \ln(T) \cdot RCC - 9849.268 \cdot Sr + 2.929 \cdot \ln(T) + 0.036 \cdot RCC - 18.333 \quad [3\text{-}335]$$

where T = dimensionless travel time,

RCC = Regulatory Compliance Concentration, and

Sr = residual saturation.

The dimensionless travel time variable is a function of source length, groundwater velocity, longitudinal dispersivity, and decay rate.

Similarly, the results of the multi-stage backward step regression yield the following equation for estimating the time of remediation for MTBE at a clean concentration of 20 μg/L:

$$TOR_{MTBE} = -0.087 \cdot Sr \cdot \ln(T) \cdot RCC + 78.829 \cdot Sr \cdot \ln(T) + 0.546 \cdot Sr \cdot RCC - 0.004 \cdot \ln(T) \cdot RCC - 493.217 \cdot Sr + 4.132 \cdot \ln(T) + 0.026 \cdot RCC - 25.825 \quad [3\text{-}364]$$

The R^2 values for $TOR_{Benzene}$ and TOR_{MTBE} are 0.9894 and 0.9834, respectively. It is interesting to note that for $TOR_{Benzene}$ it is possible to estimate TOR without using the input variables ln(T)RCC or RCC, as it was statistically determined that neither parameter was statistically significantly different than zero. However, to preserve the overall structure of the model for both benzene and MTBE, the coefficients remain in the formulation as they are statistically significant for TOR_{MTBE}.

The results illustrated in Figure 3-115 indicate a strong agreement between the estimated values of time of remediation calculated for benzene and the TORs as determined by NAS. The figure includes all estimates for each of the RCCs (5, 2, 10 and 50 μg/L). The plot of the residuals versus the predicted TOR values, as shown in Figure 3-126 indicate non-constant variance. However, this does not diminish the capacity of the formula to adequately estimate TOR.

A similar set of plots are shown in Figures 3-137 and 3-184. They illustrate the agreement between the estimated time of remediation for MTBE and the true values as determined by NAS. The plot in Figure 3-173 represents all RCCs (20, 13, 40, and 100 μg/L). Also, Figure 3-184 shows evidence of non-constant variance. This does not reduce the effectiveness of the TOR formula as an estimating tool..

Caution should be used in estimating especially low values of time of remediation. The model is especially insensitive to values of time of remediation less than 1.5 and 2.5 years for benzene and MTBE, respectively. Furthermore, these sensitivities are observed across all RCCs and irrespective of length, where residual saturation decreases and groundwater velocity increases. This lack of sensitivity is predominant where the RCC, length and residual saturation are lowest combined with the groundwater velocity at its highest. As a result calculations of benzene and MTBE times of remediation of less than 1.5 and 2.5 years, respectively, should be used with extreme prejudice. Similarly, caution must also be exercised where TOR values are estimated in excess of 100-150 years to remediation.

Furthermore, where values for time of remediation are calculated to be negative, the values should be disregarded and it is safe to assume that the time of remediation for benzene will be less than or equal to 3 remediation years and, for MTBE, less than or equal to 0.3 remediation year. These values occur almost exclusively at a length equal to 12 m and a velocity equal to 0.75 m/d.

3.4 CONCLUDING STATEMENT

The statistical framework can be applied to a number of contaminant problems as evidenced in application to the LNAPLs – Benzene and MTBE. It has been shown that a statistical framework can be utilized to calculate the timeframe necessary to achieve acceptable regulatory goals via MNA.

The use of such a simplified mechanism to solve an otherwise complex set of equations provides an additional tool for companies, regulators, citizens and affected stakeholders to better understand one aspect of a dynamic engineering problem. A highly robust computer model with potentially hundreds of inputs has been chiseled into one equation consisting of a handful of inputs.

Higher degrees of variability of estimates are observed at extreme values of Sr, L, and V which correspond to very low and/or very high TORs. The time coefficient, T, helps to capture the combined hydrodynamic effects that are characteristic of the Peclet Number – the ratio of the rate of advective contaminant transport to the rate of diffusive contaminant transport, and the Damkohler Number – the ratio of the reaction rate to that of the flow velocity. It is noted that TOR results that are less than 5 years and greater than 100 years, while applicable, must be discussed within the proper remedial action strategy/objective context.

Caution should be used in estimating especially low values of time of remediation. The model is especially insensitive to values of time of remediation less than 1.5 and 2.5 years for benzene and MTBE, respectively. Furthermore, these sensitivities are observed across all RCCs and irrespective of length, where residual saturation decreases and groundwater velocity increases. This lack of sensitivity is predominant where the RCC, length and residual saturation are lowest combined with the groundwater velocity at its highest. As a result calculations of benzene and MTBE times of remediation of less than 1.5 and 2.5 years, respectively, should be used with extreme prejudice. Similarly, caution must also be exercised where TOR values are estimated in excess of 100-150 years to remediation.

Furthermore, where values for time of remediation are calculated to be negative, the values should be disregarded and it is safe to assume that the time of remediation for benzene will be less than or equal to 3 remediation years and, for MTBE, less than or equal to 0.3 remediation year. These values occur almost exclusively at a length equal to 12 m and a velocity equal to 0.75 m/d.

These results lay the foundation for providing effective solutions to remediation needs. These formulae are exploited in the following analysis to advance an effective mechanism for incorporating economics into the remedial decision making process.

3.5 CHAPTER THREE TABLES

TABLE 3-1: NAS PARAMETERS

Contaminated Aquifer Thickness (m)	3
Decay Rate (d^{-1})	0.002
Dispersivity Ratio (m/m)	20
Effective Porosity	0.25
Fraction of Organic Carbon	0
Hydraulic Gradient (m/m)	0.01
Longitudinal Dispersivity (m)	0.5
NAPL Source Width (m)	27
Single Zone Natural Attenuation Capacity (m^{-1})	0.072
Total Porosity	0.25

TABLE 3-2: NAS REGRESSION VARIABLES

NAPL Source Length (m)	Groundwater Velocity (m/d)	Residual Saturation	RCC (µg/L)	
			Benzene	MTBE
12	0.025	0.01	2	13
18	0.05	0.02	5	20
24	0.1	0.035	10	40
30	0.2	0.04	50	100
36	0.4	0.05	-	-
42	0.75	0.07	-	-
-	-	0.1	-	-
-	-	0.14	-	-
-	-	0.2	-	-

TABLE 3-3: TOR VALUES IN YEARS FOR A BENZENE RCC OF 5 µg/L

RESIDUAL SATURATION	Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
		Length = 12 m						Length = 18 m					
	0.01	14.2	8.0	4.2	2.1	1.1	0.6	17.3	9.9	5.2	2.7	1.4	0.7
	0.02	26.2	14.8	7.9	4.1	2.1	1.1	31.9	18.2	9.7	5.0	2.6	1.4
	0.035	44.2	24.9	13.3	6.9	3.5	1.9	53.7	30.6	16.4	8.5	4.3	2.3
	0.04	50.4	28.4	15.1	7.8	4.0	2.1	61.1	34.8	18.6	9.6	4.9	2.6
	0.05	62.3	35.1	18.7	9.6	4.9	2.6	75.7	43.1	23.1	11.9	6.1	3.3
	0.07	86.6	48.8	25.9	13.4	6.8	3.6	105.0	59.8	32.0	16.6	8.4	4.5
	0.1	122.9	69.1	36.7	18.9	9.6	5.2	148.9	84.8	45.4	23.5	11.9	6.4
	0.14	171.3	96.4	51.2	26.4	13.4	7.2	207.6	118.2	63.2	32.7	16.6	9.0
	0.2	243.8	137.2	72.8	37.6	19.1	10.3	295.5	168.3	90.0	46.6	23.7	12.7
		Length = 24 m						Length = 30 m					
	0.01	20.0	11.5	6.2	3.2	1.6	0.9	22.5	13.1	7.1	3.7	1.9	1.0
	0.02	36.9	21.3	11.4	5.9	3.0	1.6	41.4	24.2	13.1	6.8	3.5	1.9
	0.035	62.2	35.8	19.3	10.0	5.1	2.7	69.7	40.6	22.0	11.5	5.9	3.2
	0.04	70.5	40.7	21.9	11.4	5.8	3.1	79.1	46.2	25.0	13.0	6.7	3.6
	0.05	87.5	50.4	27.1	14.1	7.2	3.9	98.0	57.2	31.0	16.1	8.2	4.4
	0.07	121.1	69.9	37.6	19.5	10.0	5.4	135.9	79.3	42.9	22.4	11.4	6.2
	0.1	171.9	99.1	53.3	27.7	14.1	7.6	192.6	112.4	60.9	31.7	16.2	8.7
	0.14	239.5	138.2	74.3	38.6	19.7	10.6	268.5	156.6	84.8	44.2	22.6	12.2
	0.2	341.0	196.7	105.8	54.9	28.0	15.1	382.1	222.9	120.7	62.9	32.1	17.3
		Length = 36 m						Length = 42 m					
	0.01	24.8	14.7	8.0	4.1	2.1	1.1	27.1	16.1	8.8	4.6	2.3	1.3
	0.02	45.5	26.9	14.6	7.6	3.9	2.1	49.5	29.5	16.1	8.4	4.3	2.3
	0.035	76.8	45.3	24.6	12.9	6.6	3.5	83.2	49.6	27.2	14.2	7.3	3.9
	0.04	87.0	51.4	28.0	14.6	7.5	4.0	94.5	56.3	30.8	16.2	8.3	4.5
	0.05	107.9	63.7	34.6	18.1	9.3	5.0	117.0	69.8	38.2	20.0	10.2	5.5
	0.07	149.4	88.2	48.0	25.1	12.8	6.9	161.9	96.7	52.9	27.7	14.2	7.7
	0.1	211.7	125.0	68.0	35.6	18.2	9.8	229.4	136.9	75.0	39.3	20.1	10.9
	0.14	294.9	174.2	94.8	49.6	25.3	13.7	319.6	190.8	104.4	54.7	28.0	15.1
	0.2	419.7	247.9	134.9	70.5	36.1	19.5	454.8	271.5	148.6	77.9	39.9	21.5

TABLE 3-4: TOR VALUES IN YEARS FOR AN MTBE RCC OF 20 MG/L

Velocity (m/d) →		0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
		Length = 12 m						Length = 18 m					
RESIDUAL SATURATION	0.01	4.2	2.3	1.2	0.6	0.3	0.2	4.9	2.7	1.4	0.8	0.4	0.2
	0.02	4.6	2.6	1.4	0.7	0.3	0.2	5.4	3.1	1.6	0.8	0.4	0.2
	0.035	5.3	3.0	1.6	0.8	0.4	0.2	6.4	3.6	1.9	1.0	0.5	0.3
	0.04	5.6	3.1	1.7	0.9	0.4	0.2	6.7	3.8	2.0	1.0	0.5	0.3
	0.05	6.1	3.5	1.8	0.9	0.5	0.3	7.4	4.1	2.2	1.1	0.6	0.3
	0.07	7.4	4.1	2.2	1.1	0.6	0.3	8.8	5.0	2.6	1.3	0.7	0.4
	0.1	9.0	5.1	2.7	1.4	0.7	0.4	11.0	6.2	3.3	1.7	0.8	0.5
	0.14	12.0	6.5	3.4	1.7	0.9	0.5	14.2	7.8	4.1	2.1	1.1	0.6
	0.2	15.9	8.9	4.5	2.3	1.2	0.6	18.8	10.6	5.5	2.8	1.4	0.8
		Length = 24 m						Length = 30 m					
	0.01	5.6	3.1	1.7	0.9	0.4	0.2	6.3	3.6	1.9	1.0	0.5	0.3
	0.02	6.3	3.6	1.9	1.0	0.5	0.3	7.0	4.0	2.1	1.1	0.6	0.3
	0.035	7.2	4.1	2.2	1.1	0.6	0.3	8.2	4.7	2.5	1.3	0.7	0.4
	0.04	7.6	4.4	2.3	1.2	0.6	0.3	8.8	5.0	2.7	1.4	0.7	0.4
	0.05	8.5	4.8	2.6	1.3	0.7	0.4	9.6	5.4	2.9	1.5	0.8	0.4
	0.07	10.2	5.7	3.0	1.6	0.8	0.4	11.4	6.5	3.5	1.8	0.9	0.5
	0.1	12.7	7.1	3.8	2.0	1.0	0.5	14.2	8.0	4.3	2.2	1.1	0.6
	0.14	16.3	9.1	4.8	2.5	1.3	0.7	18.2	10.3	5.5	2.8	1.4	0.8
	0.2	21.6	12.2	6.4	3.3	1.7	0.9	24.2	13.8	7.2	3.7	1.9	1.0
		Length = 36 m						Length = 42 m					
	0.01	6.9	4.0	2.1	1.1	0.6	0.3	7.5	4.3	2.3	1.2	0.6	0.3
	0.02	7.8	4.4	2.4	1.2	0.6	0.3	8.5	4.8	2.6	1.3	0.7	0.4
	0.035	9.2	5.2	2.8	1.5	0.7	0.4	10.0	5.7	3.1	1.6	0.8	0.4
	0.04	9.7	5.5	3.0	1.5	0.8	0.4	10.6	6.1	3.3	1.7	0.9	0.5
	0.05	10.6	6.1	3.3	1.7	0.9	0.5	11.5	6.6	3.6	1.9	0.9	0.5
	0.07	12.6	7.2	3.8	2.0	1.0	0.5	13.8	7.9	4.2	2.2	1.1	0.6
	0.1	15.7	8.9	4.8	2.5	1.3	0.7	17.1	9.9	5.3	2.7	1.4	0.7
	0.14	20.1	11.4	6.1	3.1	1.6	0.9	21.8	12.5	6.7	3.5	1.8	0.9
	0.2	26.7	15.2	8.0	4.2	2.1	1.1	28.9	16.6	8.8	4.6	2.3	1.2

TABLE 3-5: TOR VALUES IN YEARS FOR A BENZENE RCC OF 2 µg/L

RESIDUAL SATURATION	Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
		Length = 12 m						Length = 18 m					
	0.01	15.3	8.7	4.4	2.3	1.2	0.6	18.4	10.5	5.5	2.8	1.4	0.8
	0.02	28.1	15.9	8.4	4.4	2.2	1.2	34.0	19.3	10.3	5.4	2.7	1.5
	0.035	47.6	26.8	14.2	7.3	3.7	2.0	57.4	32.6	17.4	9.0	4.6	2.5
	0.04	54.1	30.4	16.2	8.3	4.2	2.3	65.1	37.0	19.8	10.2	5.2	2.8
	0.05	67.1	37.7	20.0	10.3	5.2	2.8	80.7	45.9	24.5	12.7	6.5	3.5
	0.07	93.1	52.3	27.8	14.3	7.3	3.9	112.1	63.7	34.0	17.6	9.0	4.8
	0.1	132.1	74.2	39.4	20.3	10.3	5.5	159.0	90.3	48.2	25.0	12.7	6.8
	0.14	184.2	103.4	54.9	28.3	14.4	7.7	221.6	125.9	67.2	34.8	17.7	9.5
	0.2	262.3	147.2	78.1	40.3	20.5	11.0	315.6	179.2	95.7	49.5	25.2	13.5
		Length = 24 m						Length = 30 m					
	0.01	21.3	12.2	6.5	3.4	1.7	0.9	23.8	13.8	7.5	3.8	2.0	1.1
	0.02	39.1	22.4	12.1	6.3	3.2	1.7	43.8	25.4	13.7	7.2	3.7	2.0
	0.035	65.9	37.9	20.4	10.6	5.4	2.9	73.8	42.9	23.2	12.1	6.2	3.3
	0.04	75.0	43.1	23.1	12.0	6.1	3.3	83.8	48.7	26.3	13.7	7.0	3.8
	0.05	92.9	53.4	28.7	14.9	7.6	4.1	103.9	60.3	32.6	17.0	8.7	4.7
	0.07	128.8	74.0	39.8	20.6	10.5	5.7	144.1	83.6	45.2	23.6	12.0	6.5
	0.1	182.8	105.0	56.4	29.3	14.9	8.0	204.3	118.6	64.1	33.4	17.0	9.2
	0.14	254.7	146.3	78.6	40.8	20.8	11.2	284.6	165.3	89.4	46.5	23.8	12.8
	0.2	362.6	208.4	111.9	58.1	29.6	15.9	405.2	235.3	127.2	66.2	33.8	18.2
		Length = 36 m						Length = 42 m					
	0.01	26.2	15.4	8.4	4.3	2.2	1.2	28.5	16.8	9.2	4.8	2.4	1.3
	0.02	48.2	28.2	15.3	8.0	4.1	2.2	52.2	30.9	16.9	8.8	4.5	2.4
	0.035	81.1	47.6	25.8	13.5	6.9	3.7	87.9	52.1	28.4	14.9	7.6	4.1
	0.04	92.0	54.0	29.4	15.3	7.8	4.2	99.8	59.1	32.3	16.9	8.7	4.7
	0.05	114.0	67.0	36.4	19.0	9.7	5.2	123.5	73.3	40.0	20.9	10.7	5.8
	0.07	158.0	92.8	50.4	26.3	13.5	7.3	171.1	101.5	55.4	29.0	14.9	8.0
	0.1	223.9	131.5	71.5	37.3	19.1	10.3	242.5	143.8	78.5	41.1	21.1	11.4
	0.14	312.1	183.3	99.6	52.0	26.6	14.3	337.8	200.4	109.4	57.3	29.3	15.8
	0.2	444.2	260.9	141.7	74.0	37.8	20.4	480.7	285.2	155.8	81.6	41.8	22.5

TABLE 3-6: TOR VALUES IN YEARS FOR A BENZENE RCC OF 10 µg/L

	Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
		Length = 12 m						Length = 18 m					
RESIDUAL SATURATION	0.01	13.5	7.6	4.0	2.0	1.0	0.6	16.4	9.4	5.0	2.6	1.3	0.7
	0.02	24.8	14.0	7.4	3.8	1.9	1.0	30.2	17.3	9.3	4.8	2.4	1.3
	0.035	41.7	23.5	12.6	6.5	3.3	1.8	50.9	29.1	15.6	8.1	4.1	2.2
	0.04	47.4	26.8	14.2	7.3	3.7	2.0	57.9	33.1	17.7	9.2	4.7	2.5
	0.05	58.9	33.2	17.7	9.1	4.6	2.5	71.7	41.0	22.0	11.4	5.8	3.1
	0.07	81.5	46.0	24.5	12.6	6.4	3.4	99.5	56.8	30.4	15.8	8.0	4.3
	0.1	115.6	65.3	34.7	17.9	9.1	4.9	141.0	80.6	43.1	22.3	11.4	6.1
	0.14	161.3	90.9	48.3	24.9	12.7	6.8	196.6	112.3	60.1	31.1	15.8	8.5
	0.2	229.6	129.5	68.8	35.5	18.0	9.7	279.9	159.9	85.6	44.3	22.6	12.1
		Length = 24 m						Length = 30 m					
	0.01	19.1	11.0	6.0	3.0	1.5	0.8	21.6	12.6	6.9	3.5	1.8	1.0
	0.02	35.0	20.3	10.9	5.7	2.9	1.6	39.5	23.1	12.5	6.5	3.3	1.8
	0.035	59.1	34.2	18.4	9.6	4.9	2.6	66.5	39.0	21.1	11.0	5.6	3.0
	0.04	67.2	38.9	20.9	10.9	5.6	3.0	75.5	44.2	24.0	12.5	6.4	3.4
	0.05	83.2	48.1	25.9	13.5	6.9	3.7	93.4	54.7	29.7	15.5	7.9	4.3
	0.07	115.3	66.7	35.9	18.7	9.5	5.1	129.6	75.9	41.1	21.5	11.0	5.9
	0.1	163.4	94.5	50.9	26.5	13.5	7.3	183.5	107.5	58.3	30.4	15.5	8.4
	0.14	227.7	131.8	71.0	36.9	18.8	10.1	255.7	149.8	81.3	42.4	21.6	11.7
	0.2	324.1	187.6	101.0	52.5	26.8	14.4	364.0	213.2	115.7	60.3	30.8	16.6
		Length = 36 m						Length = 42 m					
	0.01	23.9	14.1	7.7	4.0	2.0	1.1	26.0	15.5	8.5	4.4	2.3	1.2
	0.02	43.6	25.8	14.1	7.4	3.8	2.0	47.4	28.4	15.6	8.1	4.2	2.3
	0.035	73.3	43.4	23.7	12.4	6.3	3.4	79.6	47.7	26.1	13.7	7.0	3.8
	0.04	83.2	49.3	26.9	14.1	7.2	3.9	90.3	54.2	29.7	15.6	8.0	4.3
	0.05	103.0	61.1	33.3	17.4	8.9	4.8	111.7	67.0	36.8	19.3	9.9	5.3
	0.07	142.6	84.6	46.1	24.1	12.3	6.7	154.7	92.9	51.0	26.7	13.7	7.4
	0.1	202.0	119.9	65.4	34.2	17.5	9.4	219.1	131.6	72.2	37.9	19.4	10.5
	0.14	281.4	167.0	91.1	47.6	24.4	13.1	305.2	183.4	100.6	52.7	27.0	14.6
	0.2	400.4	237.7	129.6	67.8	34.7	18.7	434.2	260.9	143.1	75.1	38.5	20.8

TABLE 3-7: TOR VALUES IN YEARS FOR A BENZENE RCC OF 50 µg/L

Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
RESIDUAL SATURATION	Length = 12 m						Length = 18 m					
0.01	11.5	6.6	3.5	1.7	0.9	0.5	14.3	8.2	4.4	2.3	1.1	0.6
0.02	21.2	12.0	6.4	3.3	1.7	0.9	26.2	15.1	8.1	4.2	2.2	1.2
0.035	35.6	20.3	10.8	5.6	2.8	1.5	44.1	25.4	13.7	7.1	3.6	1.9
0.04	40.5	23.0	12.2	6.3	3.2	1.7	50.1	28.9	15.5	8.1	4.1	2.2
0.05	50.1	28.4	15.2	7.9	4.0	2.1	62.1	35.8	19.2	10.0	5.1	2.7
0.07	69.4	39.5	21.1	10.9	5.5	3.0	86.0	49.6	26.6	13.8	7.0	3.8
0.1	98.5	55.9	29.8	15.4	7.8	4.2	122.1	70.3	37.8	19.6	10.0	5.4
0.14	137.3	77.9	41.5	21.5	10.9	5.9	170.0	98.0	52.6	27.3	13.9	7.5
0.2	195.4	110.9	59.1	30.5	15.5	8.3	241.9	139.4	74.9	38.9	19.8	10.7
	Length = 24 m						Length = 30 m					
0.01	16.8	9.8	5.3	2.7	1.4	0.7	19.0	11.3	6.2	3.2	1.6	0.9
0.02	30.7	18.0	9.7	5.1	2.6	1.4	34.8	20.7	11.3	5.9	3.0	1.6
0.035	51.7	30.3	16.4	8.5	4.4	2.3	58.5	34.7	18.9	9.9	5.1	2.7
0.04	58.7	34.3	18.6	9.7	4.9	2.7	66.3	39.4	21.5	11.2	5.7	3.1
0.05	72.6	42.5	23.0	12.0	6.1	3.3	82.2	48.8	26.6	13.9	7.1	3.8
0.07	100.7	58.9	31.9	16.6	8.5	4.6	113.7	67.6	36.8	19.3	9.8	5.3
0.1	142.6	83.5	45.2	23.5	12.0	6.5	161.2	95.7	52.2	27.3	13.9	7.5
0.14	198.8	116.3	62.9	32.8	16.7	9.0	224.4	133.3	72.7	38.0	19.4	10.5
0.2	282.8	165.5	89.6	46.6	23.8	12.8	319.3	189.7	103.4	54.0	27.6	14.9
	Length = 36 m						Length = 42 m					
0.01	21.2	12.7	7.0	3.6	1.8	1.0	23.1	14.1	7.8	4.0	2.1	1.1
0.02	38.6	23.2	12.7	6.7	3.4	1.8	42.0	25.7	14.1	7.4	3.8	2.1
0.035	64.7	38.9	21.4	11.2	5.7	3.1	70.3	43.0	23.7	12.5	6.4	3.5
0.04	73.3	44.2	24.3	12.7	6.5	3.5	79.7	48.8	26.9	14.2	7.3	3.9
0.05	90.8	54.7	30.0	15.7	8.1	4.4	98.5	60.4	33.3	17.5	9.0	4.9
0.07	125.5	75.8	41.6	21.8	11.2	6.0	136.3	83.6	46.2	24.3	12.5	6.7
0.1	177.8	107.3	58.9	30.9	15.8	8.5	193.0	118.3	65.4	34.4	17.6	9.5
0.14	247.5	149.5	82.0	43.0	22.0	11.9	268.7	164.8	91.1	47.9	24.6	13.3
0.2	352.2	212.7	116.7	61.2	31.3	16.9	382.2	234.5	129.5	68.1	35.0	18.9

TABLE 3-8: TOR VALUES IN YEARS FOR AN MTBE RCC OF 13 µg/L

Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
RESIDUAL SATURATION	Length = 12 m						Length = 18 m					
0.01	4.2	2.4	1.3	0.6	0.3	0.2	5.0	2.8	1.5	0.8	0.4	0.2
0.02	4.7	2.7	1.4	0.7	0.4	0.2	5.6	3.1	1.7	0.9	0.4	0.2
0.035	5.4	3.1	1.6	0.8	0.4	0.2	6.5	3.7	2.0	1.0	0.5	0.3
0.04	5.7	3.2	1.7	0.9	0.4	0.2	6.8	3.9	2.1	1.1	0.5	0.3
0.05	6.4	3.6	1.9	1.0	0.5	0.3	7.5	4.3	2.3	1.2	0.6	0.3
0.07	7.5	4.2	2.2	1.1	0.6	0.3	9.0	5.0	2.7	1.4	0.7	0.4
0.1	9.3	5.2	2.8	1.4	0.7	0.4	11.1	6.3	3.3	1.7	0.9	0.5
0.14	12.2	6.6	3.5	1.8	0.9	0.5	14.6	8.0	4.2	2.2	1.1	0.6
0.2	16.3	9.2	4.7	2.4	1.2	0.6	19.3	10.8	5.6	2.9	1.5	0.8
	Length = 24 m						Length = 30 m					
0.01	5.7	3.2	1.7	0.9	0.4	0.2	6.4	3.6	1.9	1.0	0.5	0.3
0.02	6.4	3.6	1.9	1.0	0.5	0.3	7.1	4.0	2.2	1.1	0.6	0.3
0.035	7.4	4.3	2.3	1.2	0.6	0.3	8.3	4.8	2.6	1.3	0.7	0.4
0.04	7.9	4.5	2.4	1.2	0.6	0.3	8.9	5.0	2.7	1.4	0.7	0.4
0.05	8.6	5.0	2.6	1.3	0.7	0.4	9.9	5.6	3.0	1.5	0.8	0.4
0.07	10.4	5.9	3.1	1.6	0.8	0.4	11.7	6.6	3.5	1.8	0.9	0.5
0.1	12.9	7.3	3.9	2.0	1.0	0.5	14.5	8.2	4.4	2.3	1.1	0.6
0.14	16.7	9.3	4.9	2.5	1.3	0.7	18.6	10.5	5.6	2.9	1.5	0.8
0.2	22.1	12.4	6.5	3.4	1.7	0.9	24.8	14.0	7.4	3.8	1.9	1.0
	Length = 36 m						Length = 42 m					
0.01	7.1	4.1	2.2	1.1	0.6	0.3	7.7	4.4	2.4	1.2	0.6	0.3
0.02	7.9	4.5	2.4	1.2	0.6	0.3	8.6	4.9	2.7	1.4	0.7	0.4
0.035	9.3	5.3	2.8	1.5	0.8	0.4	10.2	5.9	3.1	1.6	0.8	0.4
0.04	9.9	5.6	3.0	1.6	0.8	0.4	10.7	6.2	3.3	1.7	0.9	0.5
0.05	10.8	6.2	3.3	1.7	0.9	0.5	11.8	6.8	3.6	1.9	1.0	0.5
0.07	12.9	7.3	3.9	2.0	1.0	0.6	14.1	8.0	4.3	2.2	1.1	0.6
0.1	16.0	9.2	4.9	2.5	1.3	0.7	17.5	10.0	5.4	2.8	1.4	0.8
0.14	20.5	11.7	6.2	3.2	1.6	0.9	22.4	12.7	6.8	3.5	1.8	1.0
0.2	27.2	15.5	8.2	4.2	2.2	1.2	29.5	17.0	9.0	4.7	2.4	1.3

TABLE 3-9: TOR VALUES IN YEARS FOR AN MTBE RCC OF 40 µg/L

Velocity (m/d) →		0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
		Length = 12 m						Length = 18 m					
RESIDUAL SATURATION	0.01	3.9	2.2	1.2	0.6	0.3	0.2	4.7	2.6	1.4	0.7	0.4	0.2
	0.02	4.5	2.4	1.3	0.7	0.3	0.2	5.2	2.9	1.6	0.8	0.4	0.2
	0.035	5.0	2.9	1.5	0.8	0.4	0.2	6.1	3.4	1.8	0.9	0.5	0.3
	0.04	5.4	3.0	1.6	0.8	0.4	0.2	6.4	3.6	2.0	1.0	0.5	0.3
	0.05	5.8	3.3	1.7	0.9	0.5	0.2	7.1	4.0	2.1	1.1	0.6	0.3
	0.07	7.0	3.9	2.1	1.1	0.5	0.3	8.5	4.8	2.5	1.3	0.7	0.4
	0.1	8.6	4.8	2.6	1.3	0.7	0.4	10.4	5.9	3.1	1.6	0.8	0.4
	0.14	11.4	6.1	3.3	1.7	0.8	0.5	13.6	7.5	4.0	2.0	1.0	0.6
	0.2	15.2	8.5	4.3	2.2	1.1	0.6	18.1	10.1	5.3	2.7	1.4	0.7
		Length = 24 m						Length = 30 m					
	0.01	5.4	3.0	1.6	0.8	0.4	0.2	6.0	3.4	1.8	0.9	0.5	0.3
	0.02	6.0	3.4	1.8	0.9	0.5	0.3	6.8	3.8	2.1	1.1	0.5	0.3
	0.035	7.1	4.0	2.1	1.1	0.6	0.3	7.9	4.5	2.4	1.3	0.6	0.3
	0.04	7.5	4.3	2.3	1.2	0.6	0.3	8.5	4.8	2.6	1.3	0.7	0.4
	0.05	8.2	4.7	2.5	1.3	0.6	0.3	9.3	5.3	2.8	1.5	0.7	0.4
	0.07	9.7	5.5	2.9	1.5	0.8	0.4	11.0	6.3	3.3	1.7	0.9	0.5
	0.1	12.1	6.9	3.6	1.9	1.0	0.5	13.8	7.8	4.2	2.2	1.1	0.6
	0.14	15.7	8.7	4.6	2.4	1.2	0.7	17.7	9.9	5.3	2.7	1.4	0.7
	0.2	20.7	11.7	6.2	3.2	1.6	0.9	23.4	13.3	7.0	3.6	1.8	1.0
		Length = 36 m						Length = 42 m					
	0.01	6.7	3.8	2.0	1.1	0.5	0.3	7.2	4.2	2.2	1.2	0.6	0.3
	0.02	7.5	4.3	2.3	1.2	0.6	0.3	8.2	4.7	2.5	1.3	0.7	0.4
	0.035	8.7	5.1	2.7	1.4	0.7	0.4	9.6	5.6	3.0	1.6	0.8	0.4
	0.04	9.3	5.4	2.9	1.5	0.8	0.4	10.3	5.9	3.2	1.6	0.8	0.4
	0.05	10.3	5.9	3.1	1.6	0.8	0.4	11.3	6.5	3.5	1.8	0.9	0.5
	0.07	12.2	7.0	3.7	1.9	1.0	0.5	13.4	7.7	4.1	2.1	1.1	0.6
	0.1	15.1	8.7	4.6	2.4	1.2	0.7	16.7	9.6	5.1	2.7	1.3	0.7
	0.14	19.4	11.0	5.9	3.1	1.5	0.8	21.1	12.2	6.5	3.4	1.7	0.9
	0.2	25.7	14.7	7.8	4.0	2.0	1.1	28.0	16.1	8.6	4.4	2.3	1.2

TABLE 3-10: TOR VALUES IN YEARS FOR AN MTBE RCC OF 100 µg/L

Velocity (m/d) →	0.025	0.05	0.1	0.2	0.4	0.75	0.025	0.05	0.1	0.2	0.4	0.75
RESIDUAL SATURATION	Length = 12 m						Length = 18 m					
0.01	3.6	2.0	1.1	0.6	0.3	0.2	4.5	2.4	1.3	0.7	0.3	0.2
0.02	4.2	2.3	1.2	0.6	0.3	0.2	4.9	2.8	1.5	0.8	0.4	0.2
0.035	4.7	2.7	1.4	0.7	0.4	0.2	5.7	3.3	1.7	0.9	0.5	0.2
0.04	5.0	2.8	1.5	0.8	0.4	0.2	6.1	3.4	1.9	0.9	0.5	0.3
0.05	5.6	3.1	1.6	0.8	0.4	0.2	6.7	3.8	2.0	1.0	0.5	0.3
0.07	6.5	3.6	1.9	1.0	0.5	0.3	7.9	4.5	2.4	1.2	0.6	0.3
0.1	8.1	4.5	2.4	1.2	0.6	0.3	9.9	5.6	2.9	1.5	0.8	0.4
0.14	10.7	5.8	3.0	1.6	0.8	0.4	12.8	7.1	3.8	1.9	1.0	0.5
0.2	14.2	8.0	4.1	2.1	1.1	0.6	17.0	9.6	5.0	2.6	1.3	0.7
	Length = 24 m						Length = 30 m					
0.01	5.0	2.9	1.5	0.8	0.4	0.2	5.7	3.3	1.7	0.9	0.5	0.2
0.02	5.7	3.2	1.7	0.9	0.4	0.2	6.4	3.7	2.0	1.0	0.5	0.3
0.035	6.7	3.8	2.0	1.0	0.5	0.3	7.6	4.3	2.3	1.2	0.6	0.3
0.04	7.1	4.1	2.1	1.1	0.6	0.3	8.1	4.6	2.4	1.3	0.6	0.3
0.05	7.8	4.4	2.3	1.2	0.6	0.3	8.8	5.0	2.7	1.4	0.7	0.4
0.07	9.2	5.2	2.8	1.4	0.7	0.4	10.4	5.9	3.2	1.6	0.8	0.5
0.1	11.5	6.5	3.5	1.8	0.9	0.5	13.1	7.4	4.0	2.0	1.0	0.6
0.14	14.9	8.3	4.4	2.3	1.2	0.6	16.7	9.5	5.0	2.6	1.3	0.7
0.2	19.6	11.2	5.8	3.0	1.5	0.8	22.1	12.6	6.7	3.4	1.7	0.9
	Length = 36 m						Length = 42 m					
0.01	6.2	3.6	2.0	1.0	0.5	0.3	6.8	4.0	2.2	1.1	0.6	0.3
0.02	7.1	4.1	2.2	1.1	0.6	0.3	7.8	4.5	2.4	1.3	0.6	0.3
0.035	8.5	4.9	2.6	1.3	0.7	0.4	9.2	5.4	2.9	1.5	0.8	0.4
0.04	8.9	5.1	2.8	1.4	0.7	0.4	9.7	5.7	3.0	1.6	0.8	0.4
0.05	9.9	5.7	3.0	1.6	0.8	0.4	10.7	6.2	3.3	1.7	0.9	0.5
0.07	11.7	6.6	3.6	1.9	0.9	0.5	12.8	7.3	3.9	2.0	1.0	0.6
0.1	14.4	8.3	4.4	2.3	1.2	0.6	15.9	9.2	4.9	2.6	1.3	0.7
0.14	18.5	10.6	5.7	2.9	1.5	0.8	20.2	11.7	6.3	3.2	1.6	0.9
0.2	24.4	14.0	7.5	3.9	2.0	1.1	26.6	15.4	8.2	4.3	2.2	1.2

TABLE 3-11: TIME COEFFICIENTS AND CORRESPONDING PHASE I SLOPE AND INTERCEPT VALUES

Groundwater Velocity	Source Length	ω^*	T_D	T	Benzene		MTBE	
					a	b	a	b
0.025	12	0.9244	2.0818	999.258	1209.00	1.9873	62.423	3.1766
	18	1.3866	1.7212	1239.258	1464.60	2.5287	73.933	3.8261
	24	1.8488	1.5409	1479.258	1689.40	3.0167	85.148	4.3752
	30	2.3110	1.4327	1719.258	1892.90	3.4464	94.758	4.9873
	36	2.7732	1.3606	1959.258	2078.40	3.9522	104.400	5.5417
	42	3.2354	1.3091	2199.258	2251.20	4.4383	112.820	6.0752
0.05	12	0.4708	3.1242	749.808	680.12	1.1580	34.530	1.7931
	18	0.7061	2.4161	869.808	833.90	1.4622	41.316	2.1583
	24	0.9415	2.0621	989.808	974.72	1.7126	47.713	2.4857
	30	1.1769	1.8497	1109.808	1104.20	2.0257	53.805	2.8355
	36	1.4123	1.7081	1229.808	1227.60	2.3300	59.250	3.1665
	42	1.6477	1.6069	1349.808	1344.40	2.5778	65.029	3.4618
0.1	12	0.2376	5.2079	624.951	360.82	0.6504	17.291	1.0002
	18	0.3565	3.8053	684.951	446.13	0.7692	21.509	1.1441
	24	0.4753	3.1040	744.951	524.21	0.9222	24.861	1.3519
	30	0.5941	2.6832	804.951	597.91	1.0986	28.142	1.5428
	36	0.7129	2.4026	864.951	668.14	1.2428	31.095	1.7357
	42	0.8318	2.2023	924.951	735.86	1.4056	34.286	1.9000
0.2	12	0.1194	9.3748	562.488	186.33	0.3213	8.835	0.5027
	18	0.1791	6.5832	592.488	231.03	0.3850	10.837	0.5993
	24	0.2388	5.1874	622.488	272.16	0.4792	12.875	0.7042
	30	0.2985	4.3499	652.488	311.65	0.5617	14.273	0.8120
	36	0.3582	3.7916	682.488	349.53	0.6289	16.268	0.8868
	42	0.4179	3.3928	712.488	385.82	0.7146	18.003	0.9698
0.4	12	0.0599	17.7082	531.247	94.48	0.1858	4.822	0.2326
	18	0.0898	12.1388	546.247	117.21	0.2170	5.438	0.3093
	24	0.1197	9.3541	561.247	138.90	0.2372	6.747	0.3459
	30	0.1496	7.6833	576.247	158.98	0.3084	7.152	0.4271
	36	0.1796	6.5694	591.247	178.70	0.3297	8.149	0.4645
	42	0.2095	5.7738	606.247	197.67	0.3501	9.047	0.4982
0.75	12	0.0320	32.2916	516.666	51.04	0.0730	2.332	0.1499
	18	0.0479	21.8611	524.666	63.15	0.1004	3.190	0.1643
	24	0.0639	16.6458	532.666	74.88	0.1224	3.557	0.1816
	30	0.0799	13.5166	540.666	85.72	0.1663	3.807	0.2410
	36	0.0959	11.4305	548.666	96.81	0.1357	4.430	0.2393
	42	0.1119	9.9405	556.666	106.51	0.2082	4.540	0.2756

TABLE 3-12: CONVERSION PERCENTAGES TO ESTIMATE CLEAN CONCENTRATION TOR – BENZENE

From Benzene – 5	To Benzene - 2	To Benzene - 10	To Benzene – 50
Conversion Factor	+ 4.9%	- 4.5%	- 15.3%
Standard Deviation	1.9%	1.6%	2.2%
Estimation Error	0.8%	0.03%	0.1%
Standard Deviation	0.9%	1.0%	2.7%

TABLE 3-13: CONVERSION PERCENTAGES TO ESTIMATE CLEAN CONCENTRATION TOR – MTBE

From MTBE - 20	To MTBE - 13	To MTBE – 40	To MTBE – 100
Conversion Factor	+ 2.2%	- 3.5%	- 8.3%
Standard Deviation	0.7%	0.7%	1.3%
Estimation Error	0.1%	0.1%	0.1%
Standard Deviation	1.1%	1.3%	1.8%

3.6 CHAPTER THREE FIGURES

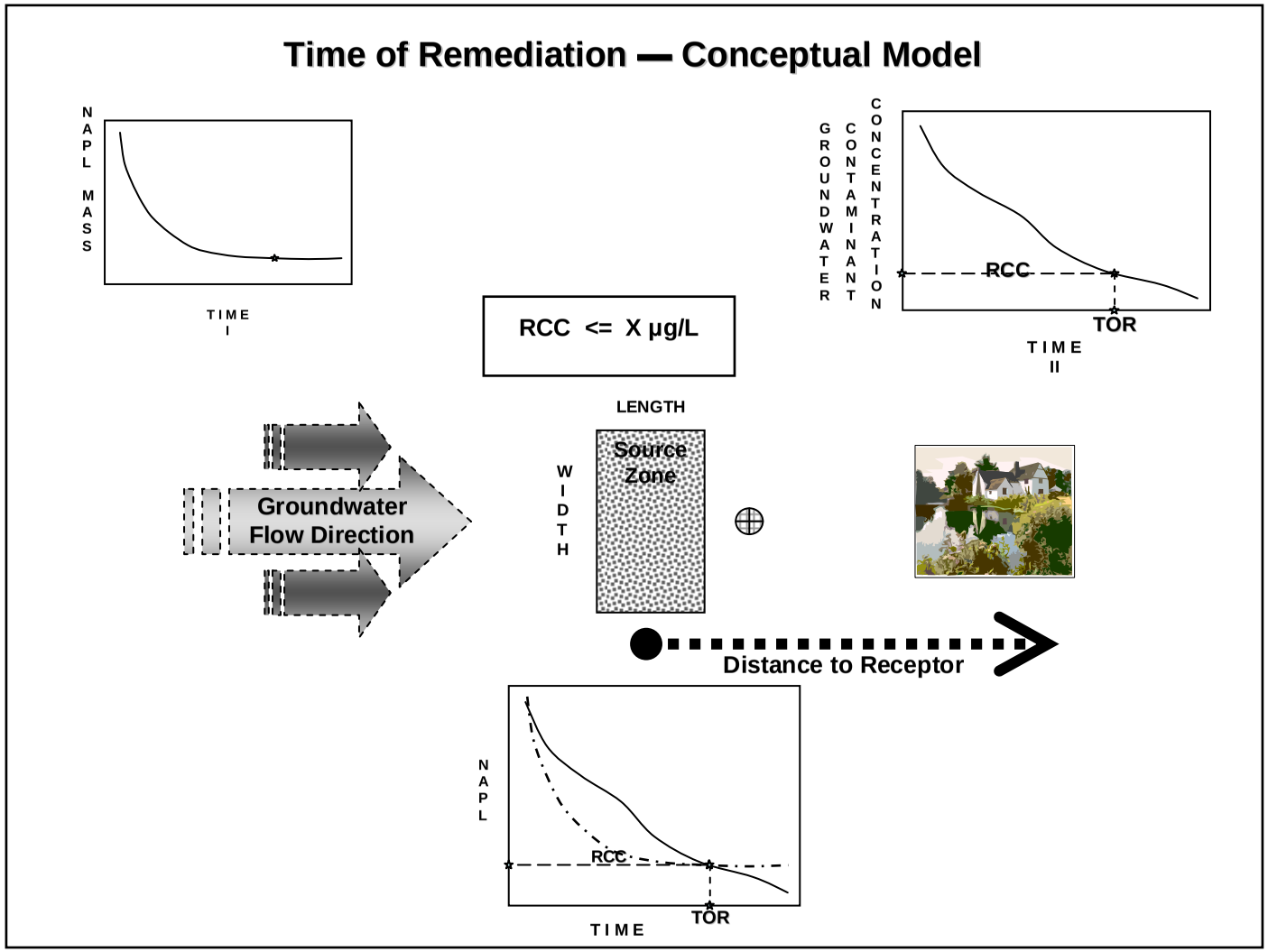
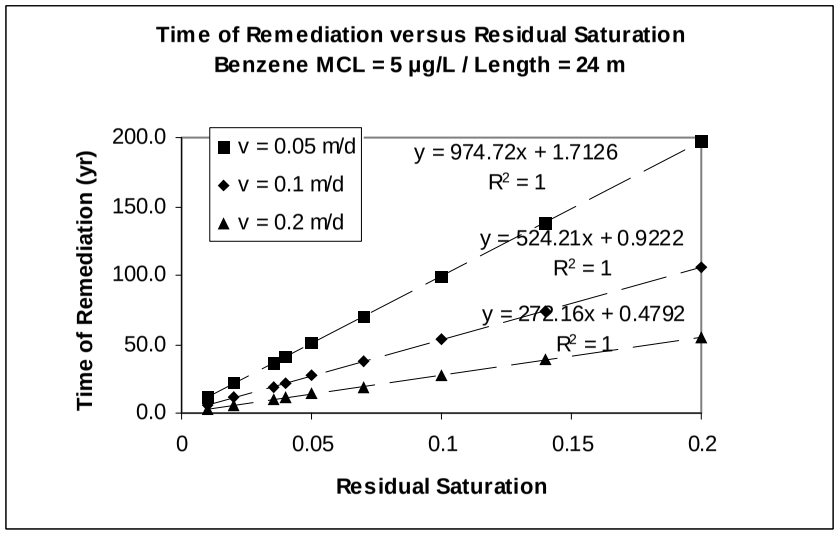
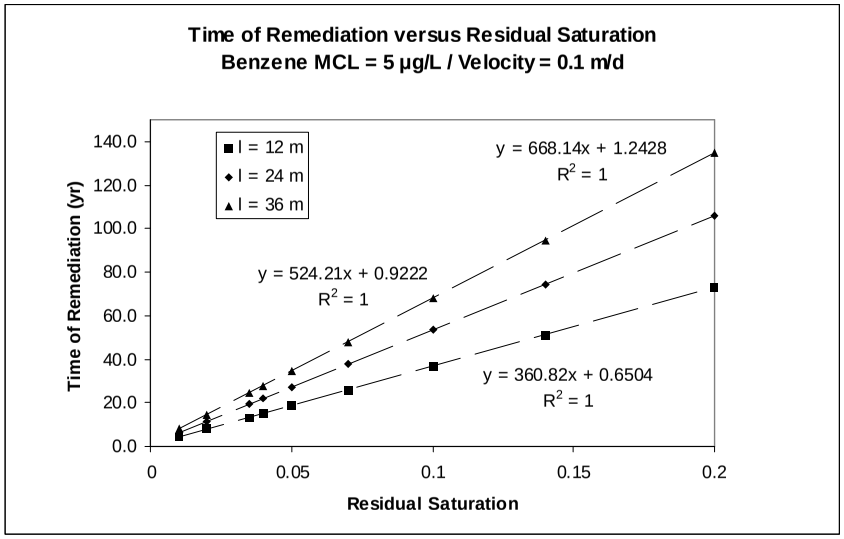
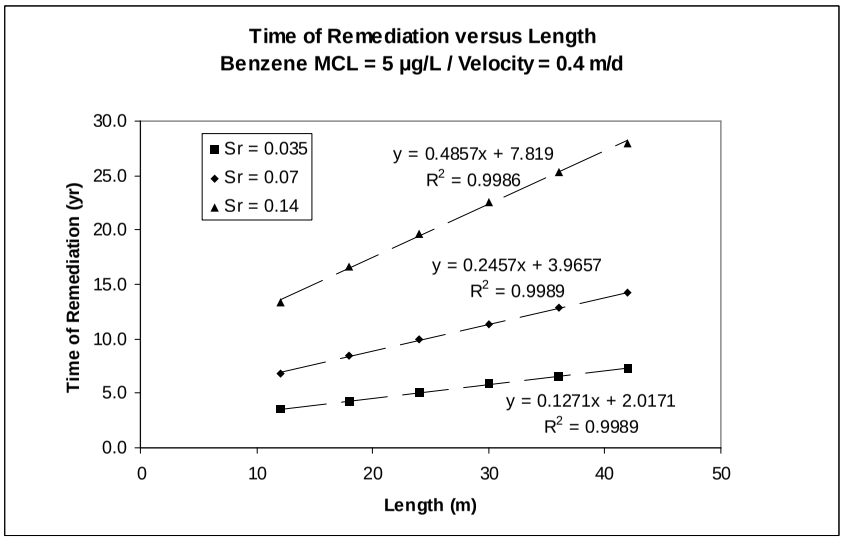


FIGURE 3-1: TIME OF REMEDIATION CONCEPTUAL MODEL







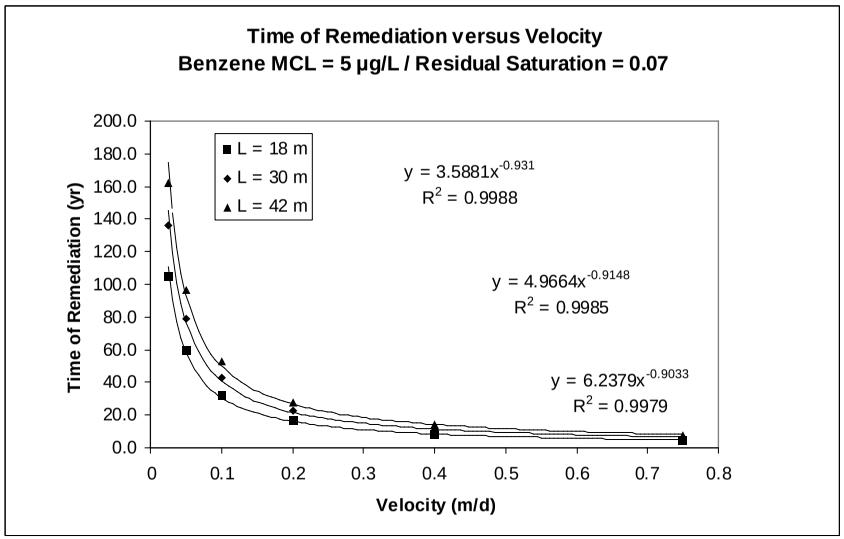
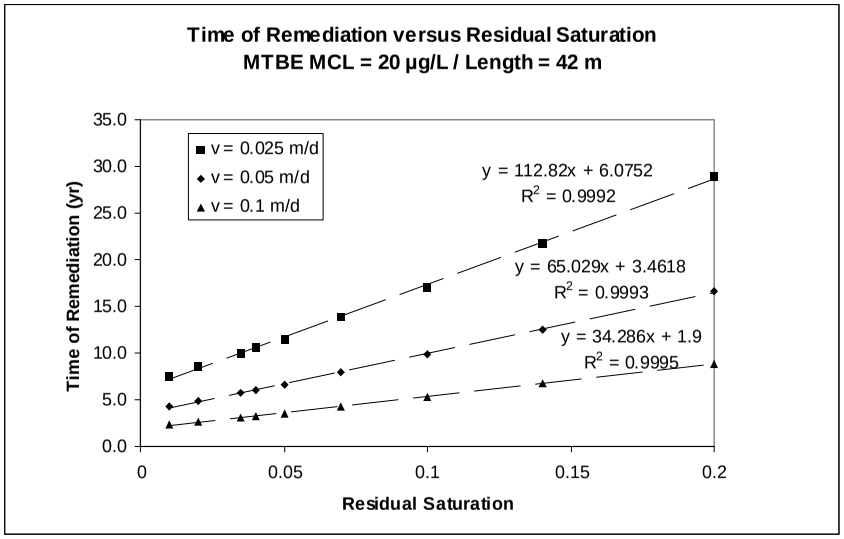
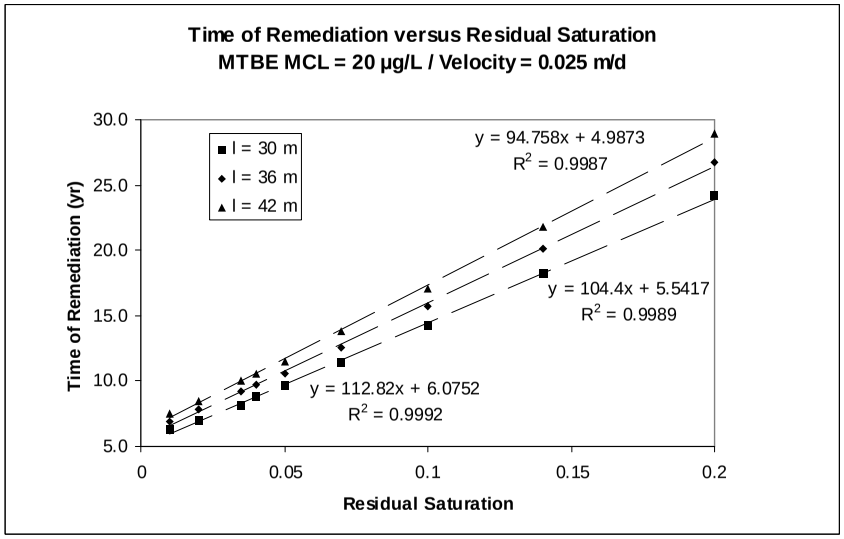
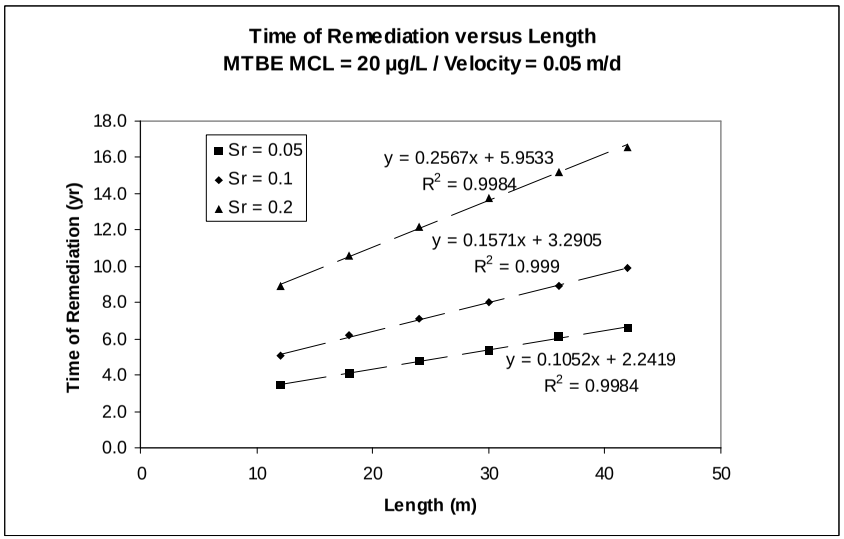


FIGURE 3-2: TIME OF REMEDIATION VERSUS INDEPENDENT VARIABLES - BENZENE







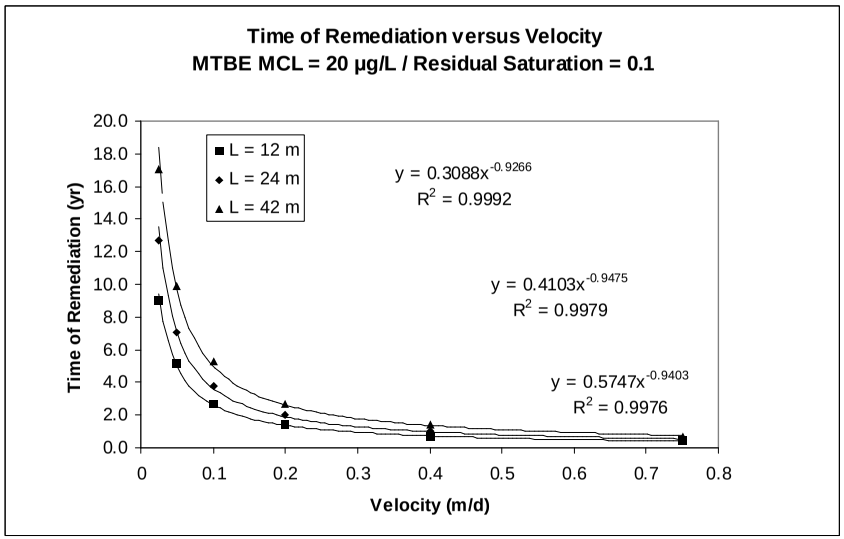


FIGURE 3-3: TIME OF REMEDIATION VERSUS INDEPENDENT VARIABLES - MTBE

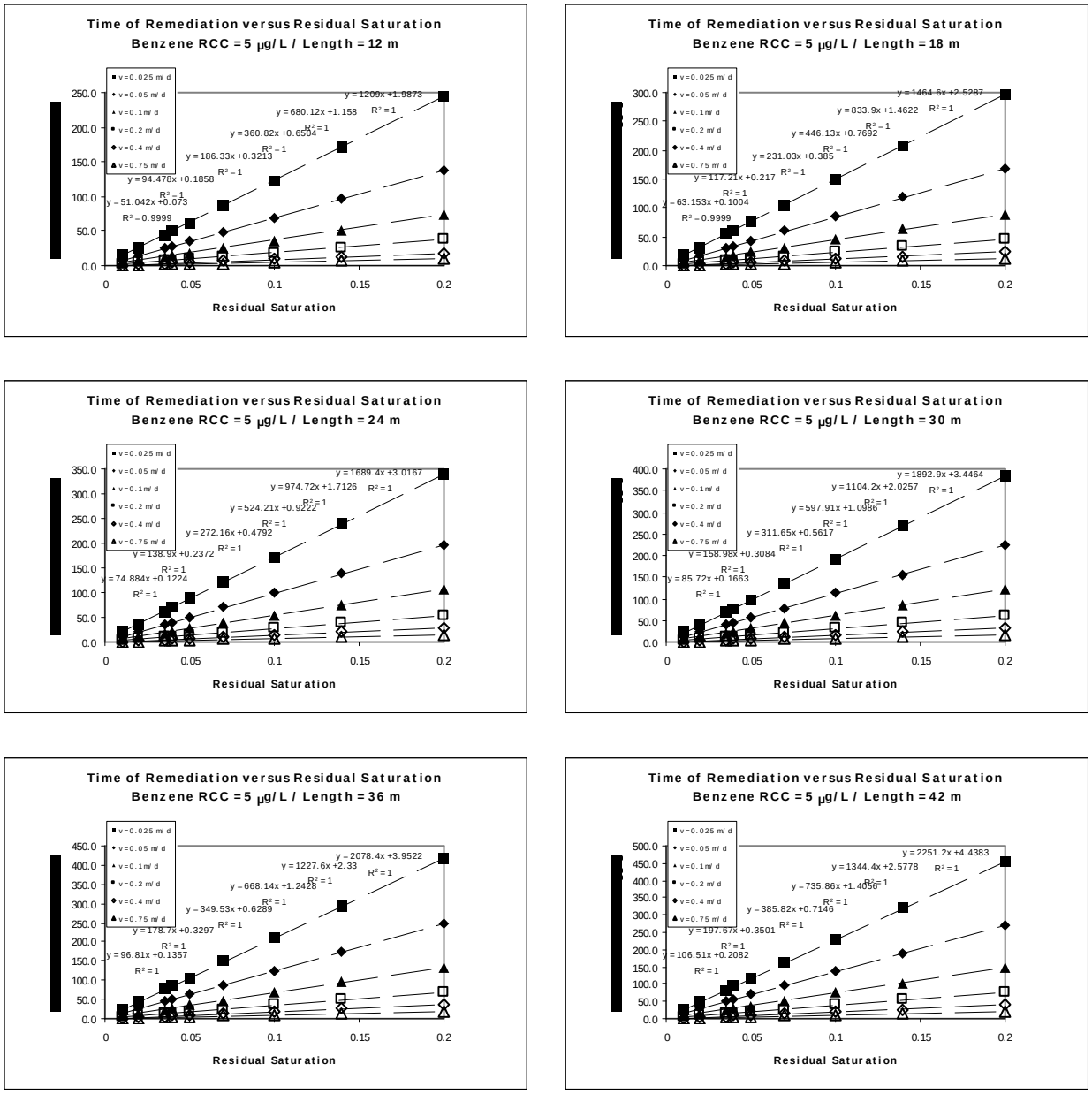


FIGURE 3-4: PHASE I REGRESSION ANALYSIS - BENZENE

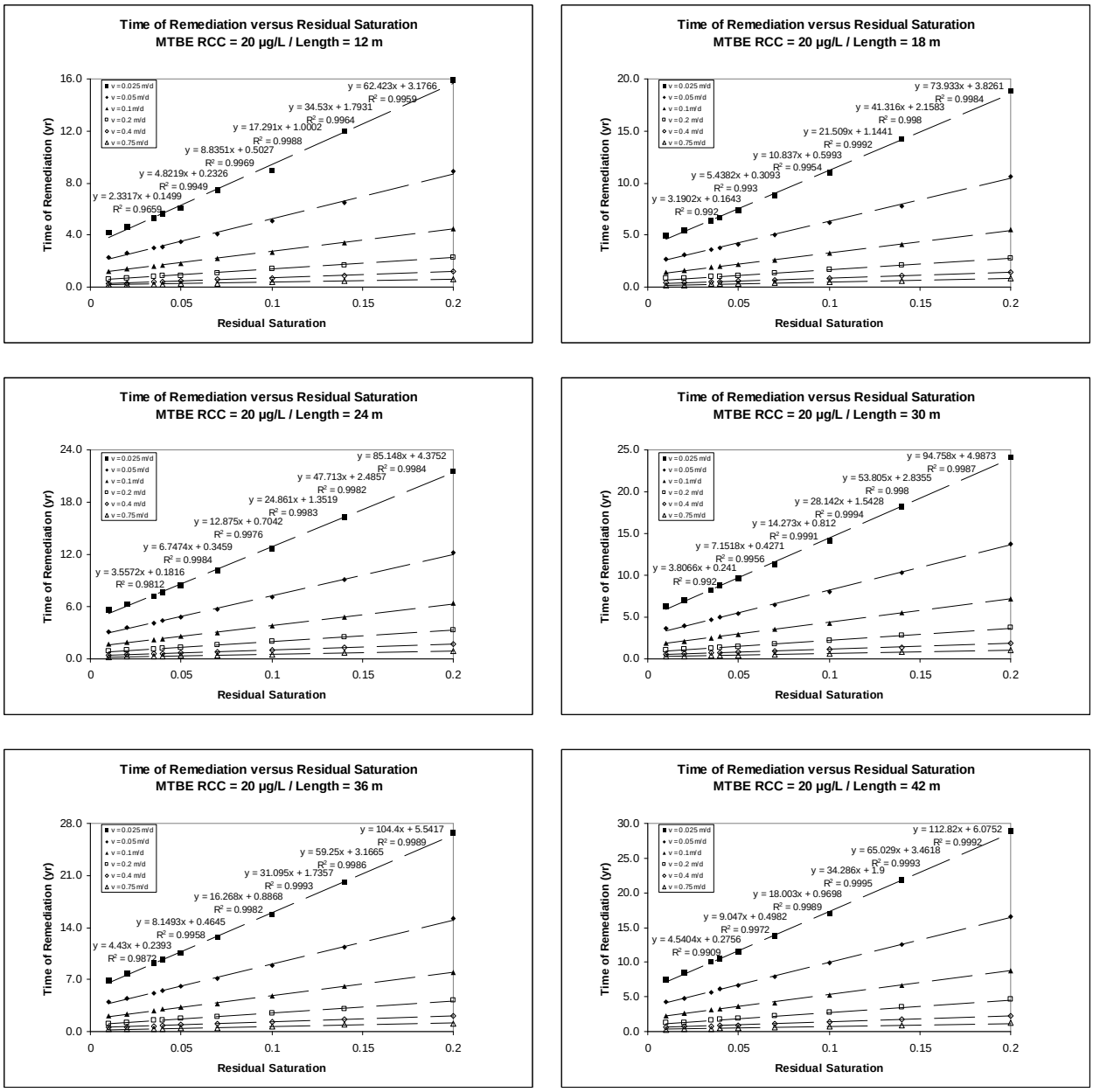
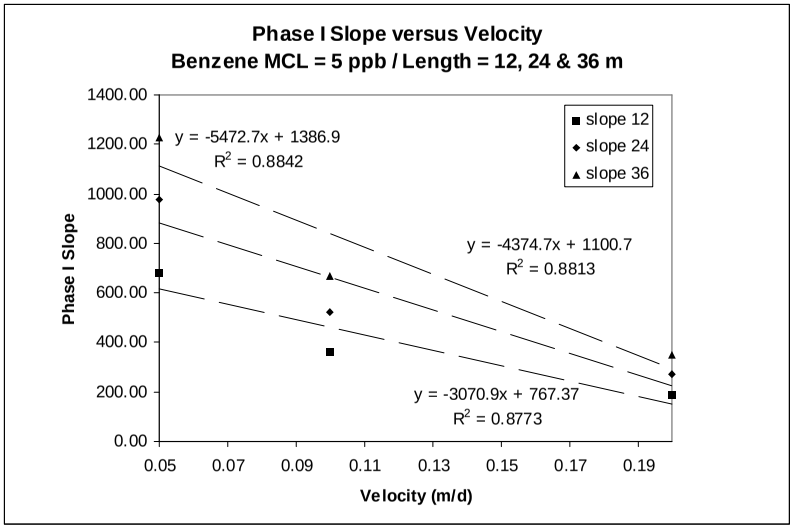
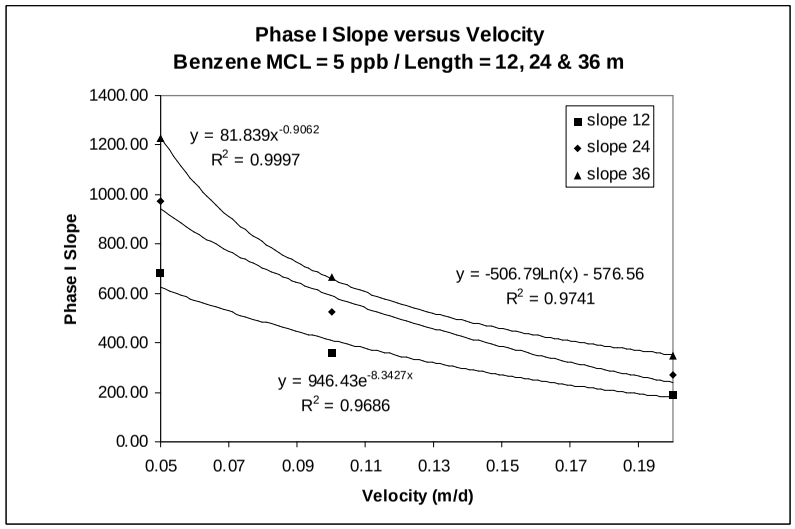
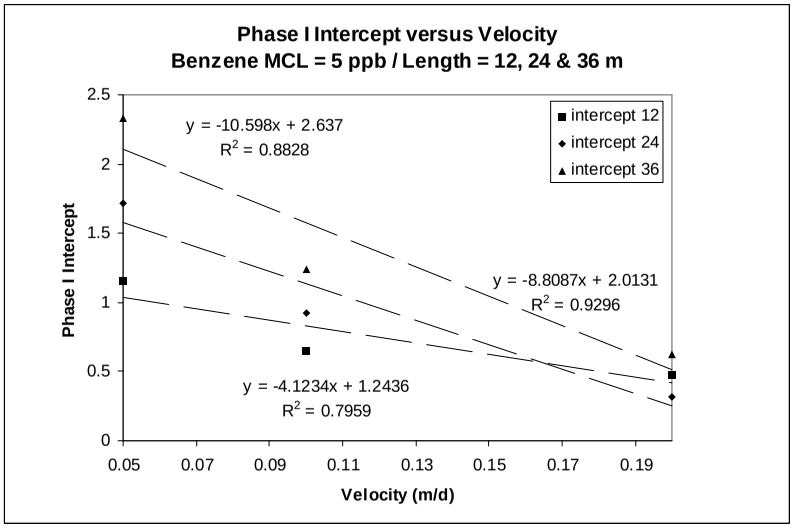


FIGURE 3-5: PHASE I REGRESSION ANALYSIS - MTBE







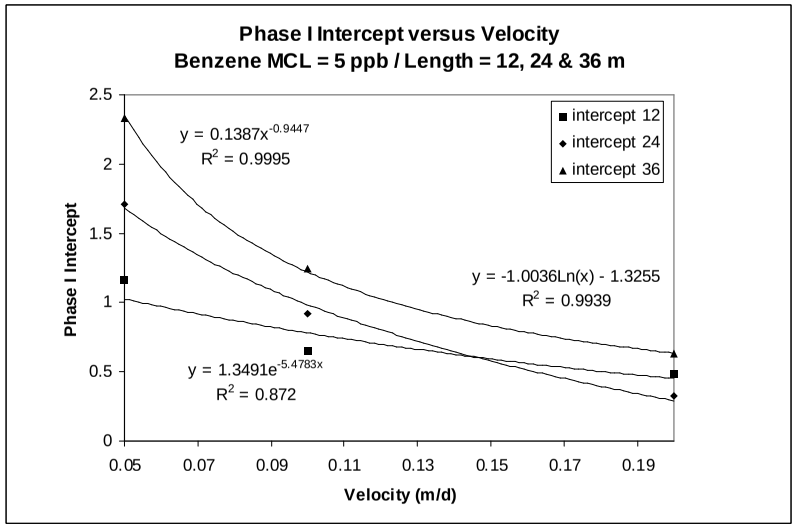


FIGURE 3-6: INITIAL PHASE II REGRESSION ANALYSIS - BENZENE

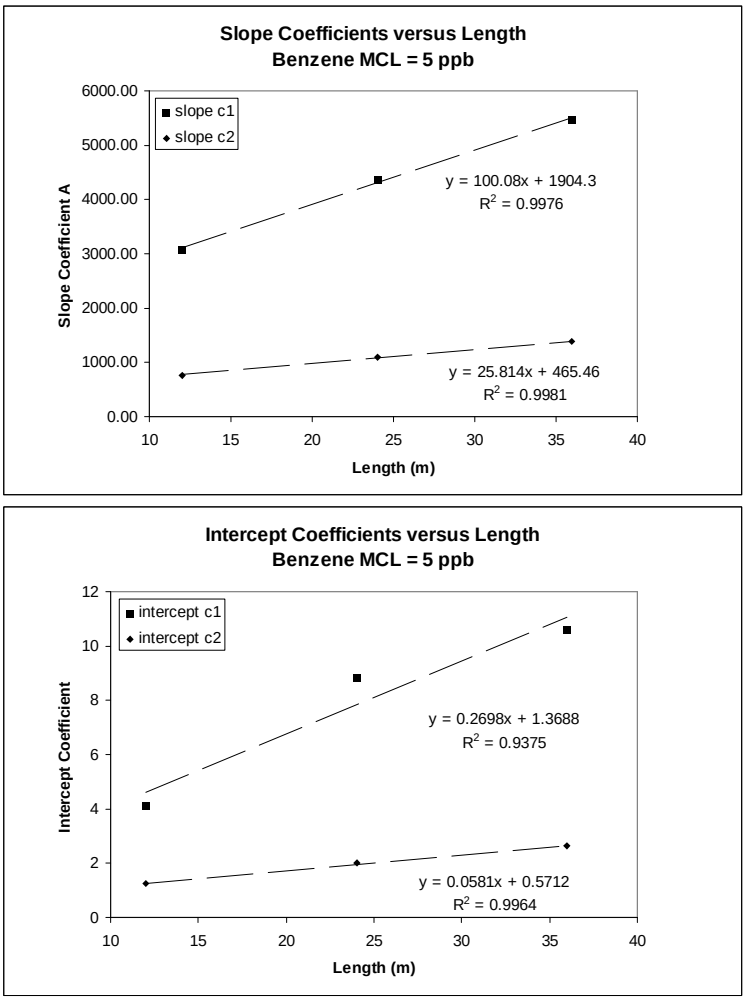


FIGURE 3-7: INITIAL PHASE III REGRESSION ANALYSIS - BENZENE

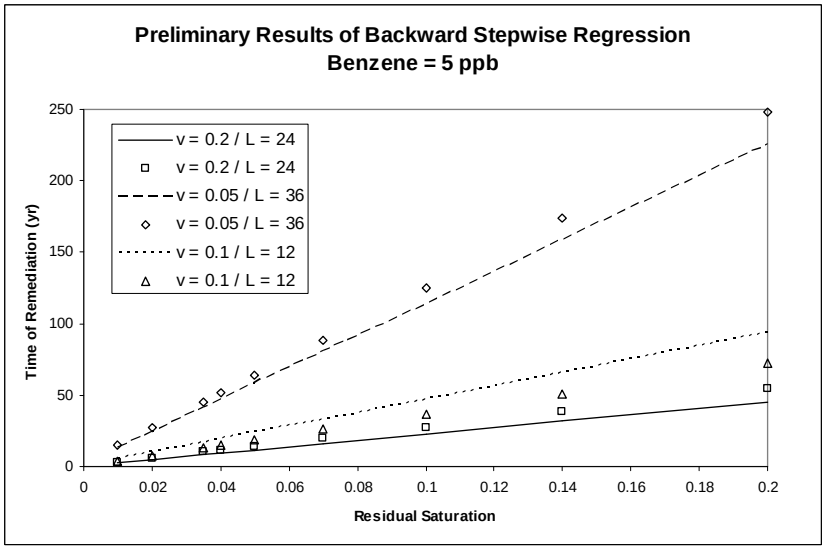


FIGURE 3-8: INITIAL PHASE COMPARISONS OF TOR RESULTS - BENZENE

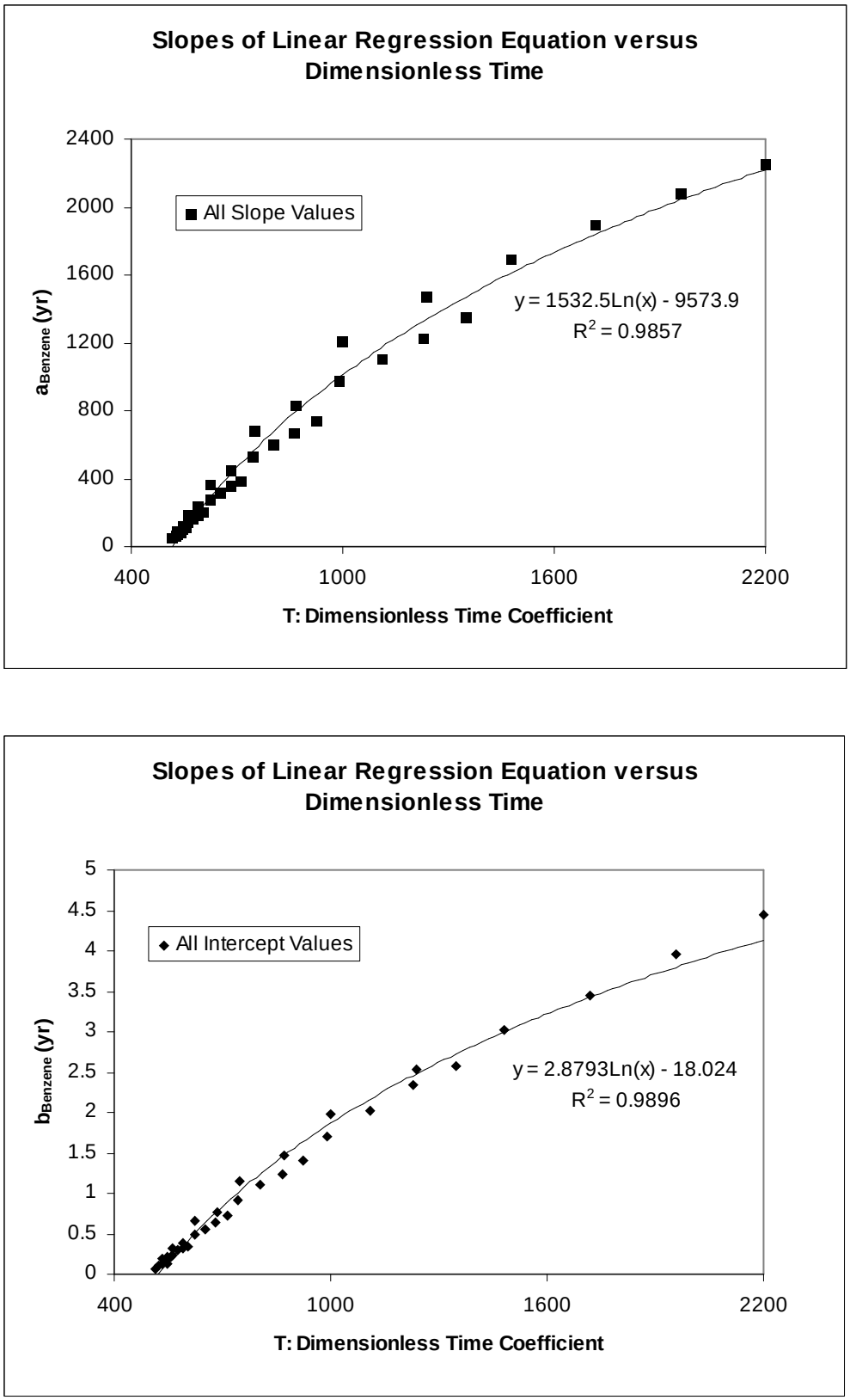
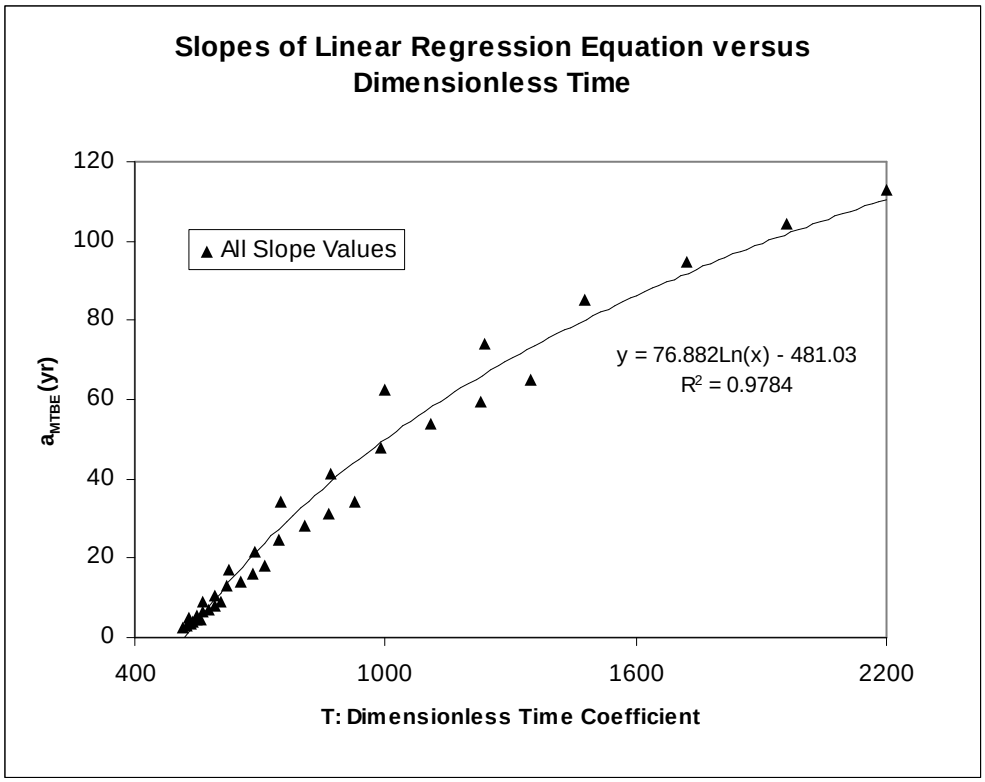


FIGURE 3-9: FINAL PHASE II ANALYSIS OF PHASE I SLOPE AND INTERCEPT VALUES VERSUS TIME COEFFICIENT – BENZENE



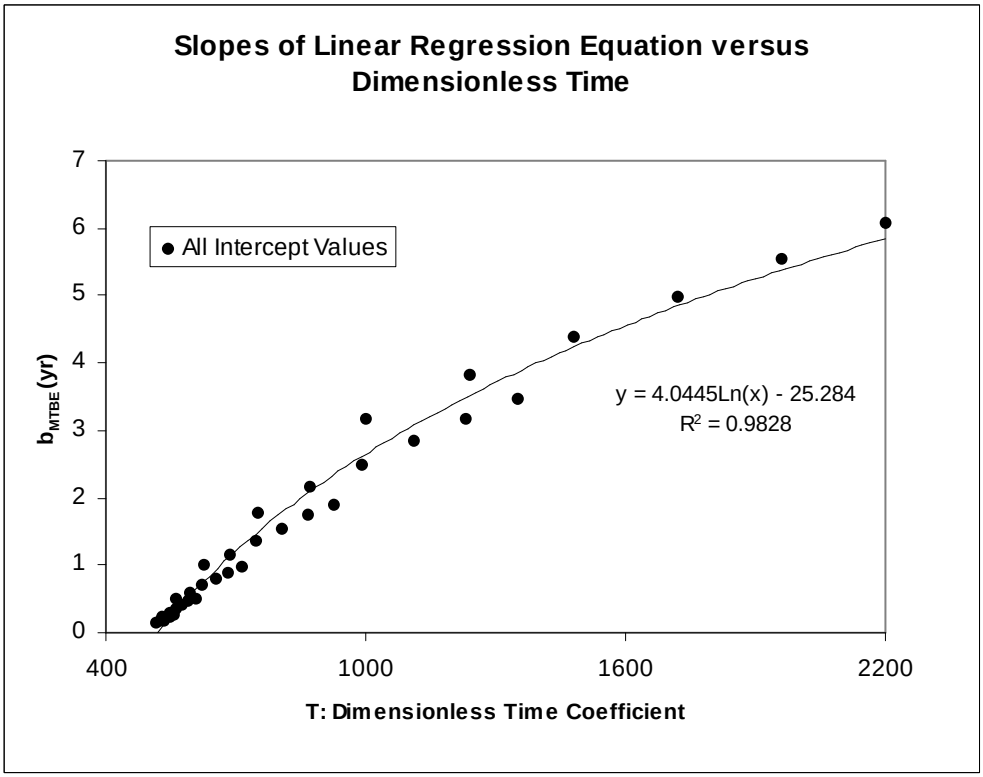


FIGURE 3-10: FINAL PHASE II ANALYSIS OF PHASE I SLOPE AND INTERCEPT VALUES VERSUS TIME COEFFICIENT – MTBE

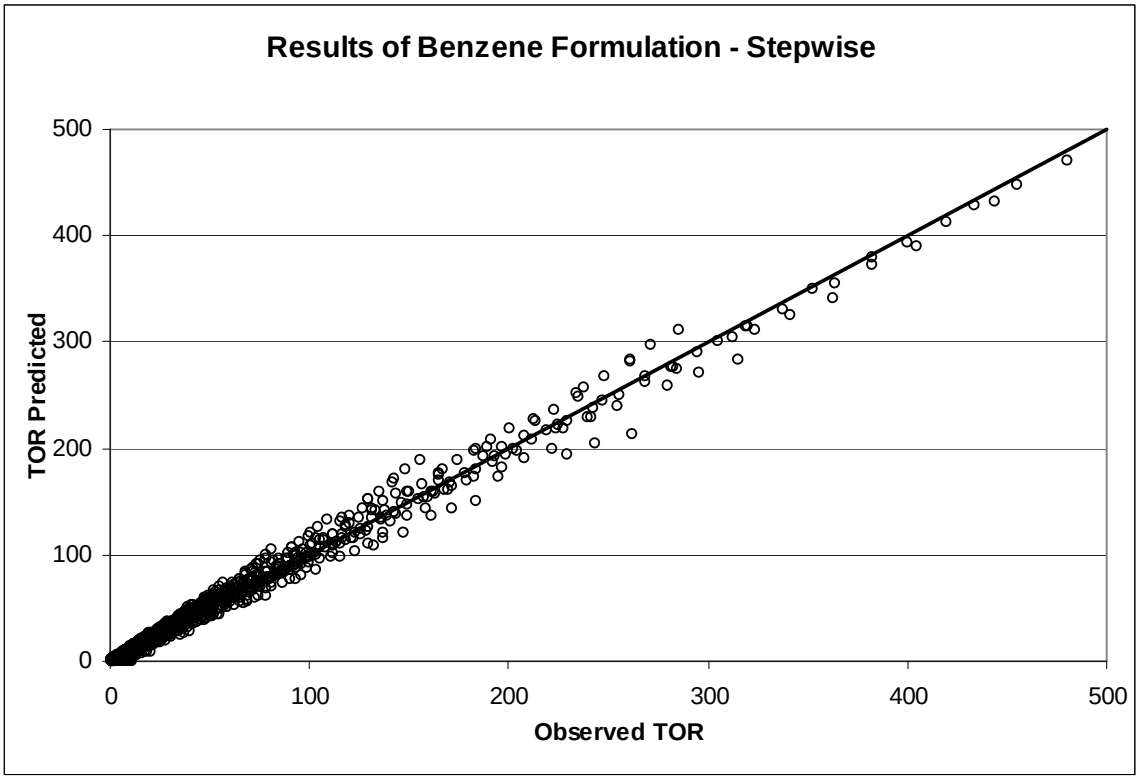


FIGURE 3-11: COMPARISON OF RESULTS – BENZENE: OBSERVED VERSUS PREDICTED

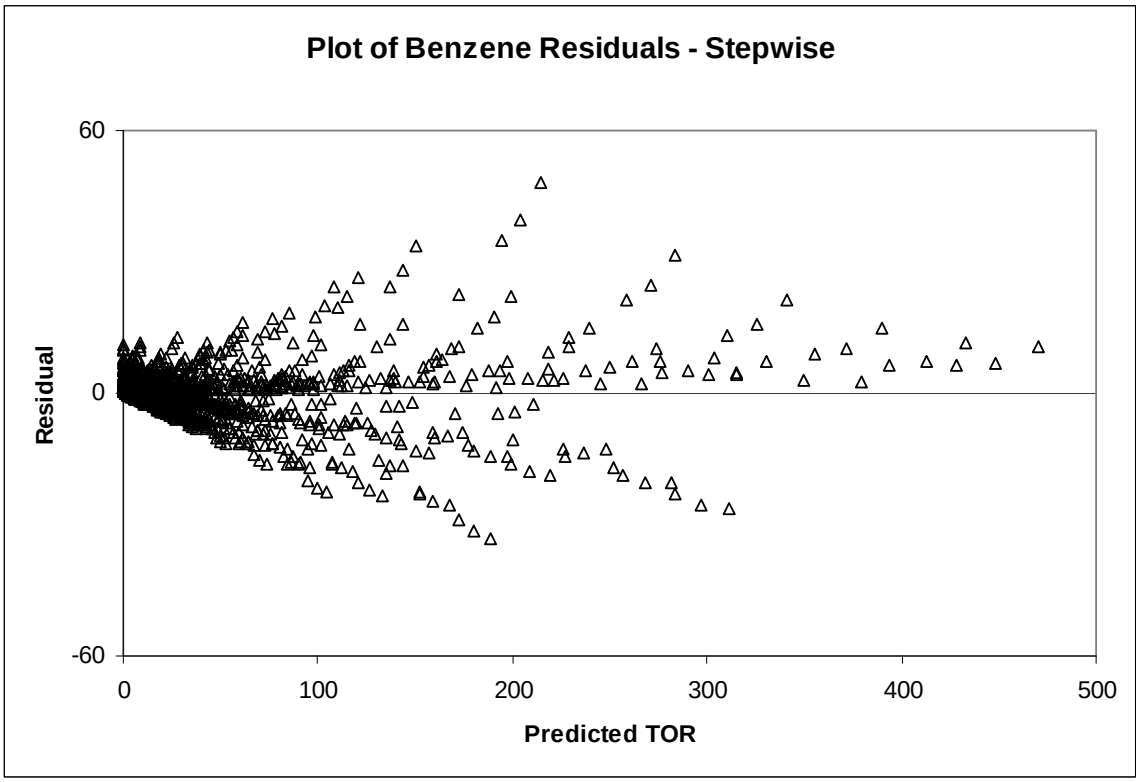


FIGURE 3-12: COMPARISON OF RESULTS – BENZENE: RESIDUALS VERSUS PREDICTED

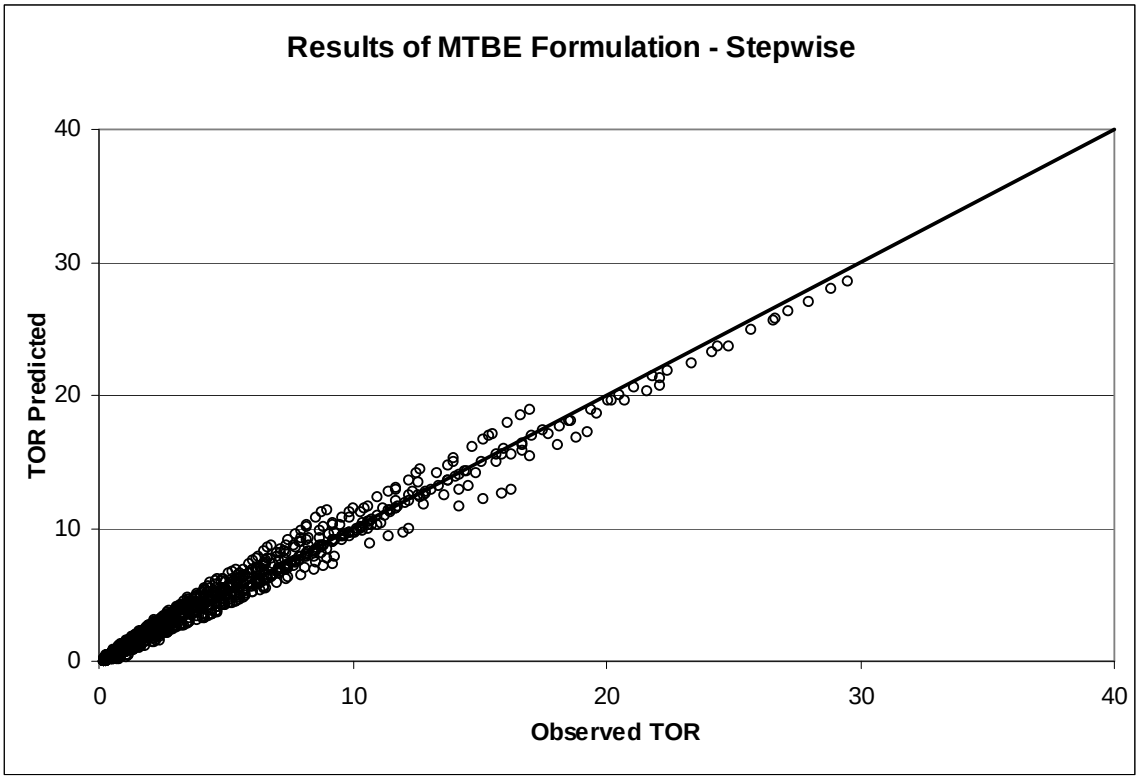


FIGURE 3-13: COMPARISON OF RESULTS – MTBE: OBSERVED VERSUS PREDICTED

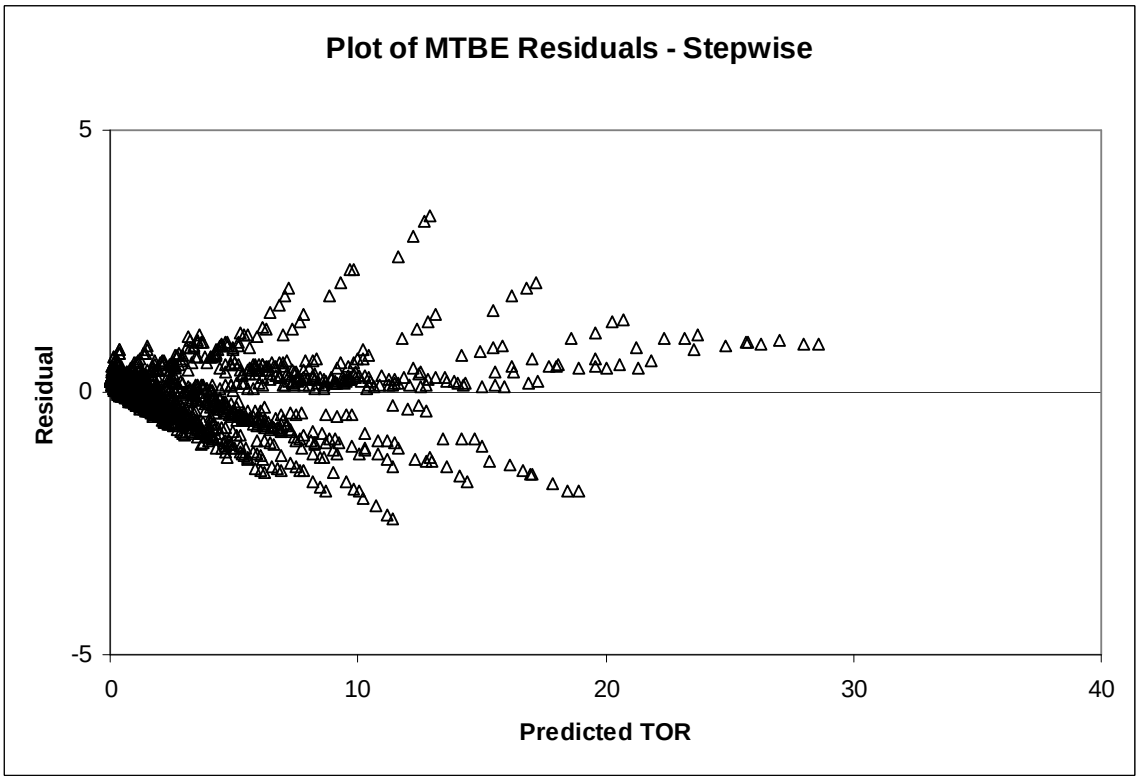


FIGURE 3-14: COMPARISON OF RESULTS – MTBE: RESIDUALS VERSUS PREDICTED

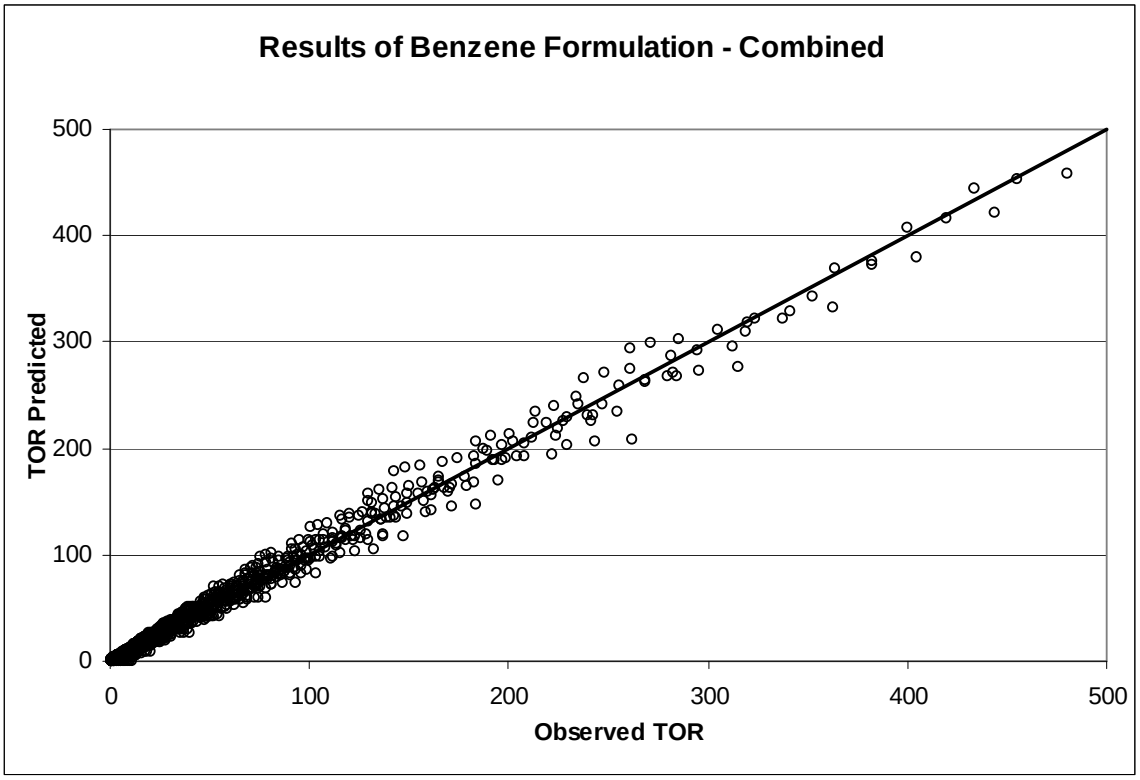


FIGURE 3-15: COMPARISON OF RESULTS – BENZENE: OBSERVED VERSUS PREDICTED

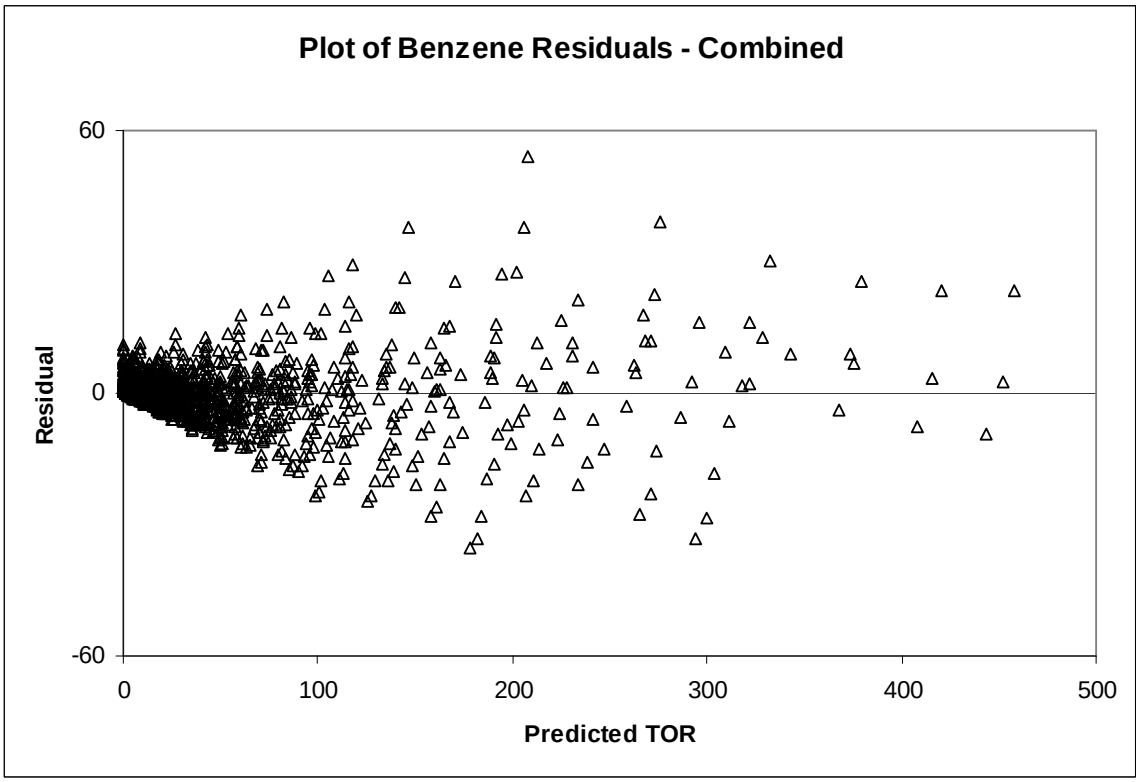


FIGURE 3-16: COMPARISON OF RESULTS – BENZENE: RESIDUALS VERSUS PREDICTED

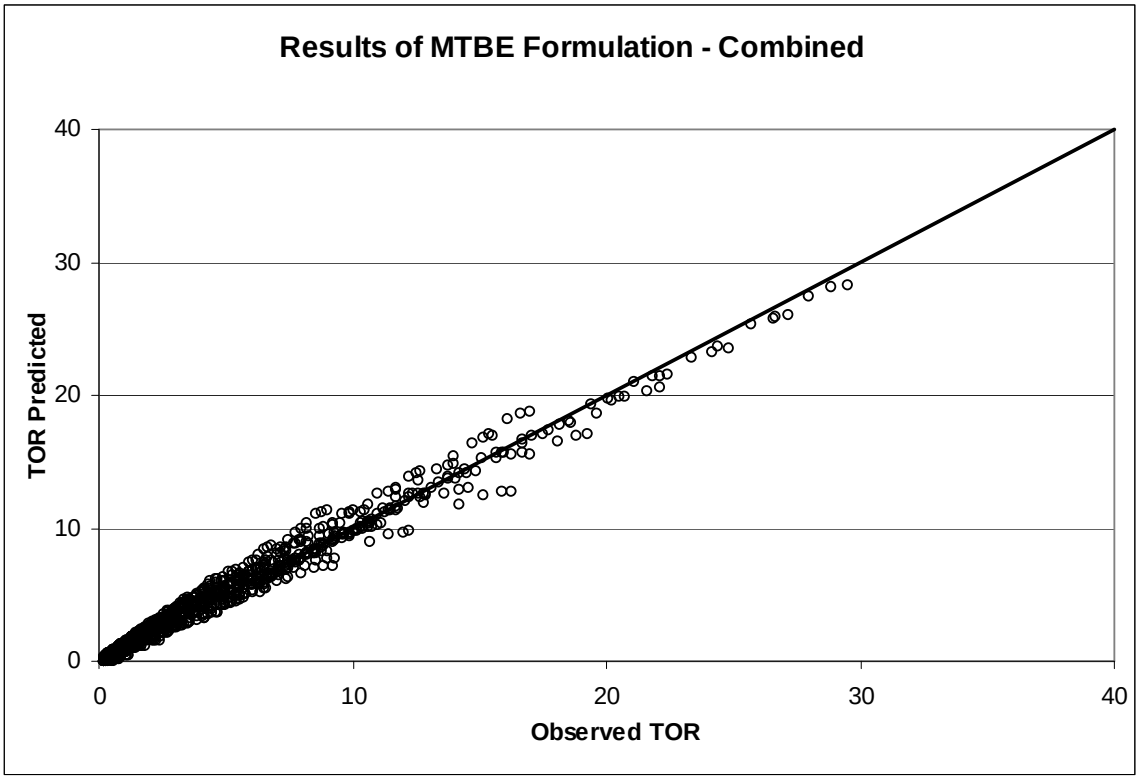


FIGURE 3-17: COMPARISON OF RESULTS – MTBE: OBSERVED VERSUS PREDICTED

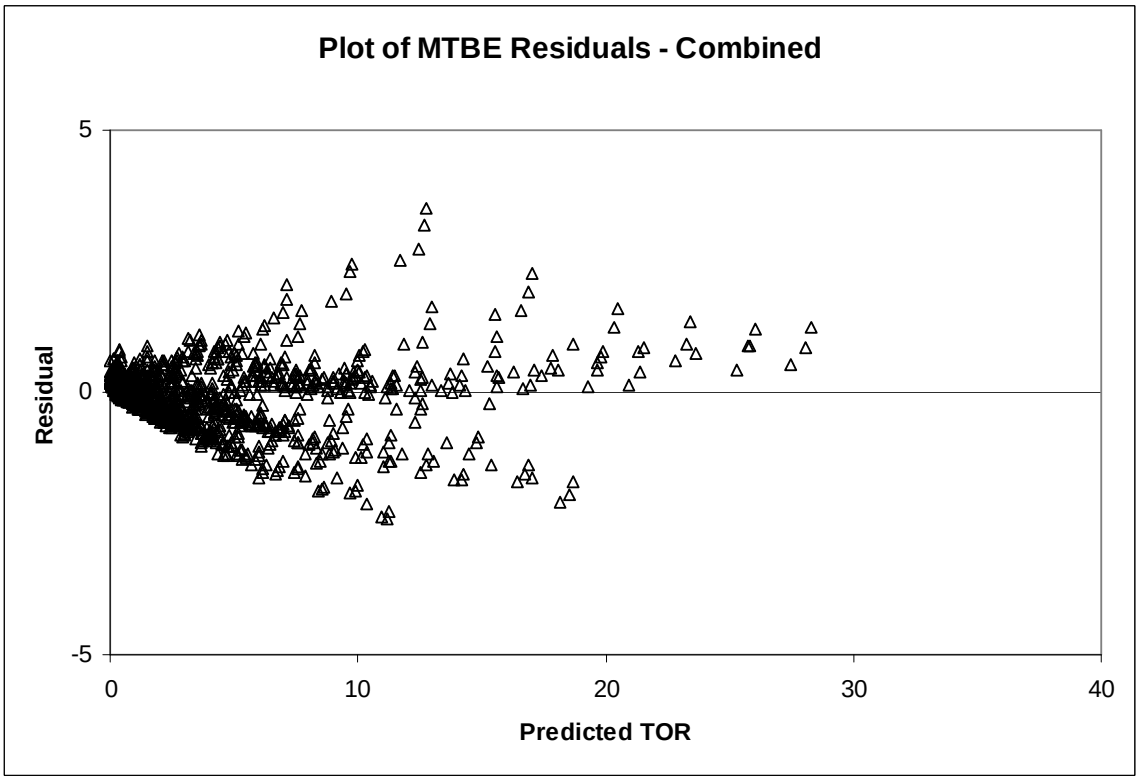


FIGURE 3-18: COMPARISON OF RESULTS – MTBE: RESIDUALS VERSUS PREDICTED

CHAPTER FOUR — EVALUATION

REMEDIAL ACTION DECISION MAKING FRAMEWORK

4.1 INTRODUCTION AND OBJECTIVES

The interdisciplinary nature of this research is highlighted by the second major research thrust. Applied science and engineering, economics, and social policy converge in the following objective of formulating an integrated framework and interactive decision tool for evaluating remedial alternatives in the context of temporal and economic considerations.

The crux of the matter then is the application of a remedial strategy that combines a source zone technology, such as excavation or SVE, with MNA. It is expected that the use of any technology other than MNA would result in an increase in costs, but most likely a decrease in the TOR. Such a tradeoff begs of a systematic framework for understanding this delicate balance that influences and is influenced by technical and non-technical drivers, alike. Thus the critical focus of this research seeks to explore the time and costs associated with each of the technologies and thus formulate an appropriate decision analysis framework to evaluate multiple source removal strategies that that reduce TOR without requiring overly excessive costs.

The framework to be developed is not intended to process a number of model inputs and result in *the* ideal remediation strategy that should be employed, such as MNA or SE only, or a combination of the two. Instead, the goal of this research is to present a legitimate decision-support tool that allows a potential business, regulator, or affected stakeholder to understand the financial and temporal effects of various approaches to site remediation. The comprehension of these factors leads to an increased awareness of the monitoring and excavation costs associated with site remediation and also, facilitate analysis of other influences that can be taken into account.

4.1 A FRAMEWORK FOR ECONOMIC OPTIMIZATION OF REMEDIAL ALTERNATIVES

Treatment technologies will differ in the costs for removing various amounts of a contaminant. In addition to costs, alternative treatment technologies affect a variable temporal component to cleanup and ultimately, closure of a specific contaminated waste site. The environmental standards

which are set mostly by government regulators must be achieved. While it is of the utmost importance that the relative benefit to receptors, human or environmental not be compromised, the considerations of time and cost will weigh heavily in the majority of remedial action decisions. As Figure 4-1 depicts, it is generally accepted that monitored natural attenuation (MNA) can be a very cost-effective remedial action strategy, especially when compared to other conventional technologies, such as excavation or soil vapor extraction (SVE) (EPA, 1999a). One of the major drawbacks however, is the time to remediation that tends to be longer than excavation and SVE applications.

It has already been documented that MNA can be an effective strategy when applied appropriately to given site-, contaminant-, and receptor-specific characteristics (EPA, 1999a). In the Final Directive from EPA's Office of Solid Waste and Emergency Response (OSWER), a cautiously measured approach is suggested when considering the use of MNA as a stand alone remedial option. The OSWER Directive leans toward the use of MNA in conjunction with a primary (active) remediation technology, constituting one component of a more holistic approach to subsurface treatment (EPA, 1999a).

This concept is illustrated in Figure 4-2 whereby the joint application of an active conventional technology such as excavation or SVE in combination with MNA would be expected to ideally reduce TOR with some expected increase in costs. This research attempts to identify the degree of source treatment and thus the corresponding extent of a complementary MNA application that achieves the remedial objective over the life of the remediation project through site/project closure. This optimal strategy is thus the target of this research – the target for relatively minimal increased costs together with relatively maximized decrease in TOR.

Remedial projects are usually projects of an urgent nature. The regulatory environment is seen as the driver, while technology is the vehicle by which we reach for the goal of protective of public health and the environment. Accordingly, the degree of protection might be described as the benefit, and the expenditures to reach this protective aim, the costs. While seldom reported in the same manner, there are also costs associated with not reaching a degree of public health or environmental protection, and certainly not reaching such a degree in a *timely* manner. The use of this framework and cost model may help establish a framework through which to apply the

principles of MNA, especially TOR estimates, to more effectively determine and implement remedial actions.

4.2 METHODOLOGY FOR AN OPTIMAL REMEDIATION STRATEGY

MNA represents a practical solution to many environmental remediation strategies. It is highly desirable for its ease of application and attractiveness of cost. Unfortunately, MNA, as a stand-alone technology, is often met with resistance from policy makers and community members because of its perceived passive approach to clean up. Essentially, it appears that the entities responsible for contaminating the environment get to *do nothing* and thus reap the benefits of having polluted without having had to clean up the resulting problem. This formulation exploits the benefits that are evident when MNA is used in combination with a source reduction strategy.

The cost model developed will look at the appropriate inputs and provide the user with a range of options to consider for future remedial action. A single technology application or conjoined technologies impact costs and time interdependently. These impacts are highly variable with respect to the remedial action objective, feasibility concerns, and temporal constraints. With this user-input-driven model, one is able to see the advantages and disadvantages to several remedial options at once and thereby proceed in the direction of greatest benefit.

4.2.1 MONITORED NATURAL ATTENUATION COST PROJECTIONS

An economic model was developed to calculate the cost of cleanup using MNA as a stand-alone remediation technology. The user is given the option to select either benzene or MTBE as the contaminant of concern. The remediation input parameters are as follows:

- Residual Saturation (Sr)
- Source Length (L)
- Groundwater Velocity (v)
- Plume Length (P_L)
- Decay Rate (k)
- Dispersivity (α).

The time of remediation calculation follows from the TOR formulations previously developed. The aforementioned input parameters lead to the calculation of travel time and subsequently, TOR. The TOR is represented for any of four pre-determined regulatory limits: Benzene – 2 ppb, 5 ppb, 10 ppb, and 50 ppb; MTBE – 13 ppb, 20 ppb, 40 ppb, and 100 ppb. A safety factor of 2 years (for all TOR values < 20 years) or 10 percent (all TOR values >= 20 years) is added to the calculated TOR estimates for use in the economic model.

While some sites may have as many as a hundred monitoring wells, it is more common to find sites with 5 – 50 wells. Therefore, this economic cost model allows for the user to view several evaluative scenarios concurrently, while highlighting the tradeoffs among technology options and temporal constraints based upon the following information shown in Table 4-1. The data collected for sites that have been or are currently undergoing remedial activities show the correlation between the length of a plume at a site and the number of wells at the site and is shown in Figure 4-3. The sites selected were drawn at random from the Electronic Document Management System (EDMS) – the online repository for all official records created or received by the Louisiana Department of Environmental Quality (LDEQ). From this information a user is able to estimate an optimal number of wells based upon a linear regression model. Alternatively, for sites with a pre-determined number of wells, that number can be input directly.

The economic input parameters that are necessary for a project life cost estimates are as follows:

- TOR
- Interest Rate (i)
- Minimum Annual Costs per Well ($W_{\$min}$)
- Average Annual Costs per Well ($W_{\$avg}$)
- Maximum Annual Costs per Well ($W_{\$max}$)
- Number of Wells (N_W).

The user has the option to employ the model-selected formulation to determine the optimal number of wells to be evaluated for a particular site scenario, or the user may input the number of wells to be factored, directly. The economic output consists of estimated lifetime net present values

for the set of wells at each MCL and for each cost estimate. Also shown are the net present values for the optimal number of wells for each MCL and cost estimate.

The determination of net present value is based upon the following formula:

$$P = A\frac{(1+i)^n - 1}{i(1+i)^n} \quad \textbf{[4-1]}$$

Here, P represents the present value of an annuity or annual expense (A) evaluated over a number of years (n) at an associated rate or interest (i). The annuity is the product of the number of wells and the annual cost associated with each well and defined according to the following:

$$A = W_{\$} * N_W \quad \textbf{[4-2]}$$

The model incorporates a significant degree of flexibility for the user as shown in the ability to input an interest (discount) rate, minimum, maximum, and average annual well costs, and the number of wells. The guidelines established by the Office of Management and Budget (OMB, 1992; OMB, 2003) recommend that a discount rate of 7 percent be used for evaluating projects, and that sensitivity analyses using rates of 5 and 9 percent be performed. It is suggested that for intergenerational projects that a lower rate (1.5 – 3 percent) could be justified.

Investigations by the United States Army Corps of Engineers (US ACE, 2005) have shown annual per well costs for long-term groundwater monitoring to be as low as $3,500 for annual monitoring of approximately 6 wells, $5,000 for semiannual monitoring, and 12,000 for quarterly monitoring a various stages of project life. For the purpose of illustrating the applicability of the model, the example case uses values of $500, $1,000, and $5,000.

4.2.2 EXCAVATION COST PROJECTIONS

Fifteen sites that used excavation as a sole remediation technology were evaluated to determine a minimum, average, and maximum cost of excavation per site. The information from these sites was made available through the LDEQ EDMS. The data collected relative to volume of soil excavated and the associated costs are presented in Table 4-2 and illustrated in Figure 4-4. The

plot of the data suggest a highly correlated linear fit to the data and thus a linear model is used to estimate the per cubic meter costs of excavation given an excavation volume. The linear model is used to determine the projected costs based upon the following equation:

$$Cost_{Excavation} = 102.61 * Volume_{SoilExcavated} + 17{,}984 \quad \textbf{[4-3]}$$

This cost equation represents the equation of the line of best fit for the data as represented in Figure 4-2, whereby the slope ($103/cu. m) of Equation 4-2 is determined by Equation 3-10 and the intercept ($18,000) is calculated according to Equation 3-11. Furthermore, maximum, average, and minimum estimates are determined as calculated from the average of the data presented in Table 4-2 and observed from the maximum and minimum values. Accordingly, the maximum, average, and minimum unit costs to excavate a site are approximately $100, $200, and $750.

The costs of excavating a site may be estimated with the knowledge of the source excavation volume, bulk density, and depth to source. The inputs necessary for this calculation include the following:

- Source Length (L)
- Source Width (W)
- Source Thickness (T_S)
- Bulk Density (D_B)
- Depth to Source (D_S)

The source volume (SV) is determined by the following:

$$SV = L * W * T_S \quad \textbf{[4-4]}$$

The source mass (SM) is determined according to

$$SM = SV * D_B \quad \textbf{[4-5]}$$

The excavation volume (EV) is determined by adding a safety excavation factor (E_{SF}) of 2.0 meters to L and W, and 1.0 meters to T_S according to the following:

$$EV = (L + 2.0) * (W + 2.0) * (D_S + T_S + 1.0) \quad \textbf{[4-6]}$$

The E_{SF} used above is a more conservative estimate of the *fluffy factor* of 1.10 as described by Kuo (1999). In the textbook, *Practical Design Calculations for Groundwater and Soil Remediation*, the fluffy factor is used to correct for the expansion of the soil post-excavation. This physical occurrence is due to in situ soil being more compact and thus denser below ground than above ground or in stockpiles. The source and excavation volumes must also account for source containers or other non-soil objects that occupy volume.

Accordingly, the excavation mass (EM) is calculated as such

$$EM = EV * D_B \quad \textbf{[4-7]}$$

From the above formulae, it is possible to estimate the projected cost to remediate a source zone at a contaminated site via excavation only:

$$Cost_{Excavation} = 103 * EV + 18{,}000 \quad \textbf{[4-8]}$$

Accordingly, the minimum unit costs to excavate a site may be approximated as \$100, the average unit costs, \$200, and the maximum unit costs, \$750, such that

$$Cost_{Minimum} = 100 * EV \quad \textbf{[4-9]}$$

$$Cost_{Average} = 200 * EV \quad \textbf{[4-10]}$$

$$Cost_{Maximum} = 750 * EV \quad \textbf{[4-11]}$$

4.2.3 EXCAVATION + MONITORED NATURAL ATTENUATION COST PROJECTIONS

The cost projections for the joint application of SE and MNA are similarly developed using the base formulation for determining TOR as previously shown. These calculations present a framework for evaluating the choice between reducing the number of remediation years and what is generally recognized as an increase in costs to facilitate such a reduction. A range of values for the amount of source volume removed ($VR_{\%}$) is established between 10 and 95percent. However a percent removal efficiency (RE%) is calculated to establish the correlation between the remaining mass and the TOR ratio as defined according to the following:

$$RE_{\%} = (100 - VR_{\%}) * (0.0095) + 0.0521 \quad \textbf{[4-12]}$$

The joint remediation years are then calculated by combining RE% and the adjusted TOR. Figure 4-5 shows the correlation plot used to estimate the TOR ratio which directly contributes to the above calculation.

4.3 RESULTS AND DISCUSSION

Consider the following three cases Alpha, Beta and Gamma that produce a low, moderate, and high TOR as given in Table 4-3. The remaining global inputs force these example cases are listed in Table 4-4. The results of the economic analysis are presented in Table 4-5 and correspond to the Figures 4-5, 4-6, and 4-7.

The results for MNA as a stand alone technology show the projected costs given a number of well scenarios. Each figure shows the projected, minimum, average, and maximum costs based upon the unit cost data derived from Table 4-2 and Equation 4-8. As expected, the cost of remediation via MNA is highly dependent upon the initial determination of TOR, which is mainly influenced by Sr, L, and v. Relative to the cases, Alpha, Beta, and Gamma, the TORs of 4, 41, and 225 years would cost approximately $60,000, 280,000, and 320,000, respectively under MNA.

The results for SE as a stand along technology show the projected costs for removing the source zone. The costs are highly dependent upon the physical size of the source zone. As the source length in the primary physical variable, it is the major factor for determining the price difference in the three test cases: Alpha – SE costs are approximately $2.3 million for an L equal to 42 m; Beta – SE costs are approximately $1.7 million for an L equal to 30 m; Gamma – SE costs are approximately $0.8 million for an L = 12 m.

An alternative method for evaluating the remedial alternatives looks at the number of remedial years saved and the corresponding increase in costs relative to those years gained. These results are shown in Table 4-6. This *marginal cost* provides an estimate of the value of foregoing an additional year of remediation. Through example Beta, it is shown that as the costs to remediate increase and the years to remediate correspondingly decrease the marginal costs per remediation year foregone decreases except for the marginal costs of 100 percent SE.

It would seem obvious that MNA would be the preferred technology for each of the cases, if in fact the decisions were solely based upon cost. However, when the TOR is factored into the decision analysis along with the costs, the tradeoffs become more subjective. Relative to case Gamma, $320,000 sounds ideal; however a TOR of 225 years is excessive and inconsistent with the guidelines adopted by EPA which mandate that a technology such as MNA reach the *protective of public health and the environment* standard in a reasonable timeframe (US EPA, 1999a).

Alternatively, if time were the only factor, SE would dominate the majority of remediation sites. Due to its relative ease of application, the costs are primarily a function of the amount of subsurface material that must be removed. Because the majority of SE projects can be completed within a year, it is ideal for timely cleanup. The major barrier is for sites that are not physically conducive to excavation and/or sites that are too massive for SE to be economical. Consider the cases, Alpha, Beta and Gamma and a decision to chose SE over MNA. While the TOR would essentially fall from 4 years to 1 year, 41 to 1, and 225 to 1, the costs would be increased by factors of 38 ($60,000 to $2.3 million), 6 ($280,000 to $1.7 million) and 2 ($320,000 to $0.8 million), respectively.

The value added of this model is in the combinatorial capacity to evaluate two technologies simultaneously, or in phased application. While EPA will allow MNA as a stand-alone technology, it is highly recommended that it be used in combination with a source treatment application. Evaluating the joint application of SE + MNA the tradeoffs are presented in Table 4-5 and illustrated in Figure 4-7. These plots show that tradeoff between the years necessary for active remediation and the associated present value costs. As expected, there is a reciprocal relationship between the costs required to remediate and the time of remediation. Figure 4-7 shows the change in costs measured against both percent volume of removed subsurface material and the number of active remediation years. These plots also include the time and costs for MNA and SE.

This model provides an assessment tool for an interested party to evaluate several stand alone technologies while also comparing the potential resource savings associated with a combined remedial approach. It is evident that the characteristics of the physical site will play an extremely important role in the cost projections. However, through this simple and user-driven spreadsheet a

more informed decision can be made relative to the impact that a single or multiple technologies will have in the short- and long-term. What Figure 4-7 also illustrates is that caution be exercised by assuming that it is always easier to simply excavate. The summary values presented for SE are the average values and because of the user-specific capability should be highly accurate. However, when moving toward the maximum boundaries, due to miscellaneous barriers and constraints, the cost of SE can rise sharply. It is suggested and highly encouraged that users conduct the due diligence such that model inputs are as realistic and site-specific as possible to ensure the appropriate use and application of results.

The choice of technology options that are deemed protective of public health and the environment often involve the considerations of TOR versus (or relative to) costs to remediate. A legitimate economic framework to evaluate single and dual technology remediation options has been developed and provides insight into the dynamics of the potential tradeoff between TOR and CTR.

Relative to cleaning up contaminated environmental sites, TOR cannot be the sole consideration, nor can the economic burden of lack thereof be the sole factor in determining a remedial agenda. It is important that difficult remediation decisions evolve through diverse constituencies of stakeholders in an interdisciplinary forum that spans the array of *affected* interests and *effecting* parties.

4.4 CHAPTER FOUR TABLES

TABLE 4-1: CONTAMINANT PLUME LENGTH AND CORRESPONDING NUMBER OF MONITORING WELLS

Plume Length (m)	Number of Wells	Plume Length (m)	Number of Wells
503	26	640	26
213	22	550	40
305	40	280	15
207	19	50	6
183	18	183	13

TABLE 4-2: COST ESTIMATES FOR EXCAVATION RELATIVE TO SOIL VOLUME REMOVED

Volume Removed (cubic meters)	Estimated Costs ($)	Estimated Costs Per Cubic Meter ($/cu. m)	Volume Removed (cubic meters)	Estimated Costs ($)	Estimated Costs Per Cubic Meter ($/cu. m)
28	$6,110	$216	7	$5,144	$748
254	$46,495	$183	1301	$157,952	$121
153	$29,852	$195	840	$98,774	$118
115	$19,445	$170	997	$126,362	$127
115	$19,465	$170	305	$78,883	$258
1413	$190,141	$135	1770	$174,233	$98
687	$126,931	$185	1668	$167,968	$101
119	$24,842	$209	-	-	-

TABLE 4-3: EXAMPLE CASE SPECIFIC INPUTS

Example Case	Alpha	Beta	Gamma
Residual Saturation	0.01	0.1	0.2
Source Length (m)	42	30	12
Groundwater Velocity (m/d)	0.75	0.2	0.025

TABLE 4-4: GLOBAL INPUTS FOR EXAMPLE CASE DEVELOPMENT

Example Case	Alpha ▪ Beta ▪ Gamma
Source Width	27
Source Thickness (m)	3
Source Zone Bulk Density	1,600
Depth to Source (m)	5
Decay Rate	0.002
Dispersivity	0.5
Plume Length	131
Plume Width	33
Plume Thickness	6
Plume Bulk Density	1,600
Interest Rate	5%
Minimum Annual Costs per Well	$500
Average Annual Costs per Well	$1,000
Maximum Annual Costs per Well	$5,000

TABLE 4-5: COMPARATIVE REMEDIAL OPTIONS

		Alpha		Beta		Gamma	
		Costs	Years To Remediate	Costs	Years to Remediate	Costs	Years to Remediate
MNA		$56,736	4	$276,710	41	$319,995	227
SE		$2,296,800	1	$1,670,400	1	$730,800	1
	Volume Removed						
SE	10%	$172,004	4	$355,695	38	$359,423	206
+	25%	$344,560	4	$469,926	32	$418,515	174
MNA	50%	$631,708	3	$647,641	22	$516,249	120
	75%	$918,211	2	$798,593	12	$600,992	66
	90%	$1,089,823	1	$872,004	5	$612,178	23

TABLE 4-6: MARGINAL COSTS PER REDUCTION IN YEARS TO REMEDIATE

	Costs to Remediate	Years to Remediate	Years Gained	Incremental Costs	Marginal Costs per Years Gained
	$56,736	4	-	-	-
A	$172,004	4	0	$115,268	N/A
L	$344,560	4	0	$172,556	N/A
P	$631,708	3	1	$287,148	$287,148
H	$918,211	2	1	$286,503	$286,503
A	$1,089,823	1	1	$171,612	$171,612
	$2,296,800	1	0	$1,206,977	N/A
	$276,710	41	-	-	-
	$355,695	38	3	$78,985	$26,328
B	$469,926	32	6	$114,231	$19,039
E	$647,641	22	10	$177,715	$17,772
T	$798,593	12	10	$150,952	$15,095
A	$872,004	5	7	$73,411	$10,487
	$1,670,400	1	4	$798,396	$199,599
	$319,995	227	-	-	-
G	$359,423	206	21	$39,428	$1,878
A	$418,515	174	32	$59,092	$1,847
M	$516,249	120	54	$97,734	$1,810
M	$600,992	66	54	$84,743	$1,569
A	$612,178	23	43	$11,186	$260
	$730,800	1	22	$118,622	$5,392

4.5 CHAPTER FOUR FIGURES

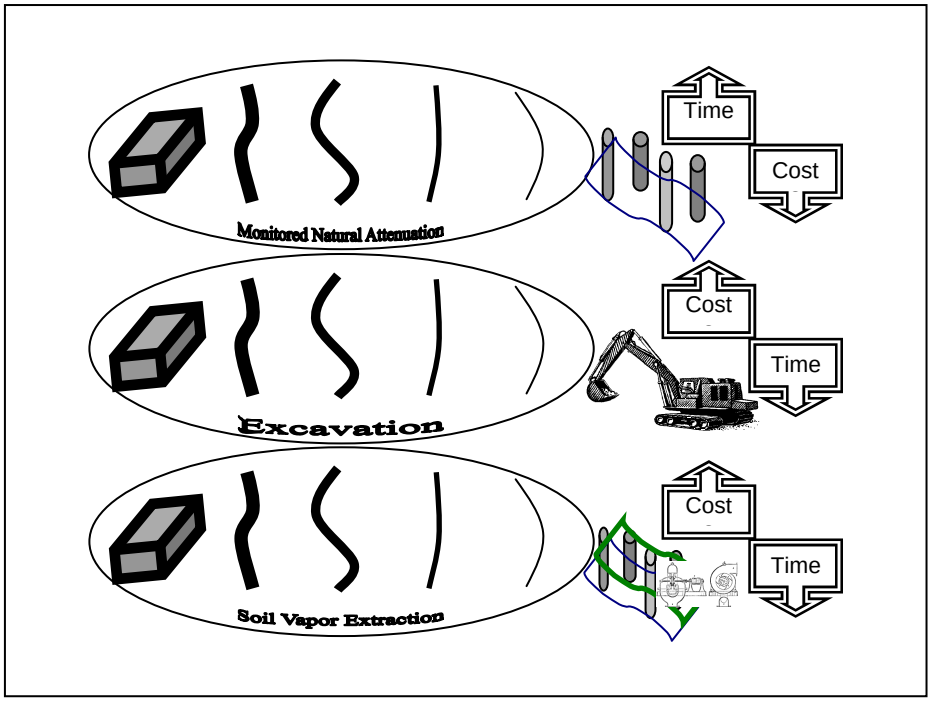


FIGURE 4-1: TEMPORAL AND COSTS CONSIDERATIONS RELATIVE TO ALTERNATIVE REMEDIATION STRATEGIES

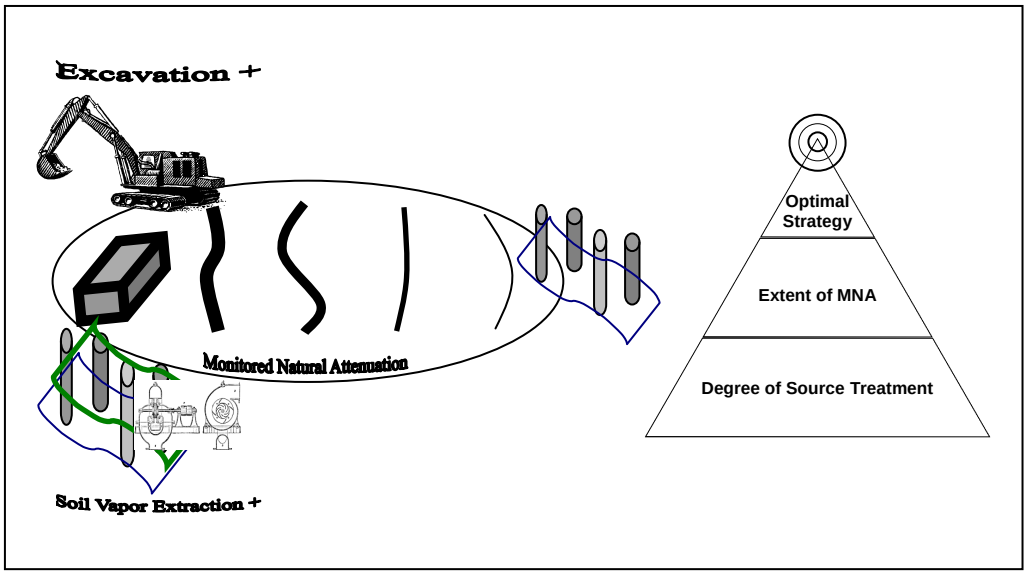


FIGURE 4-2: TEMPORAL AND COST CONCEPTUALIZATION FOR A COMBINED REMEDIAL APPROACH

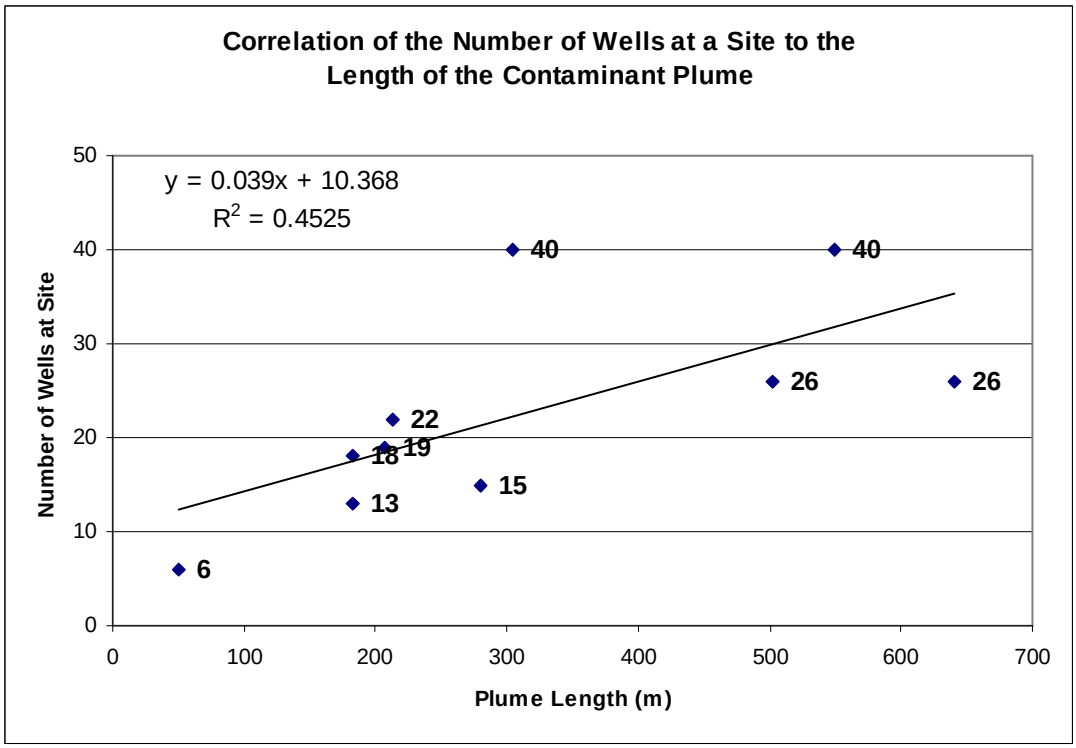


FIGURE 4-3: CORRELATION OF NUMBER OF WELLS PER SITE TO CONTAMINANT PLUME LENGTH

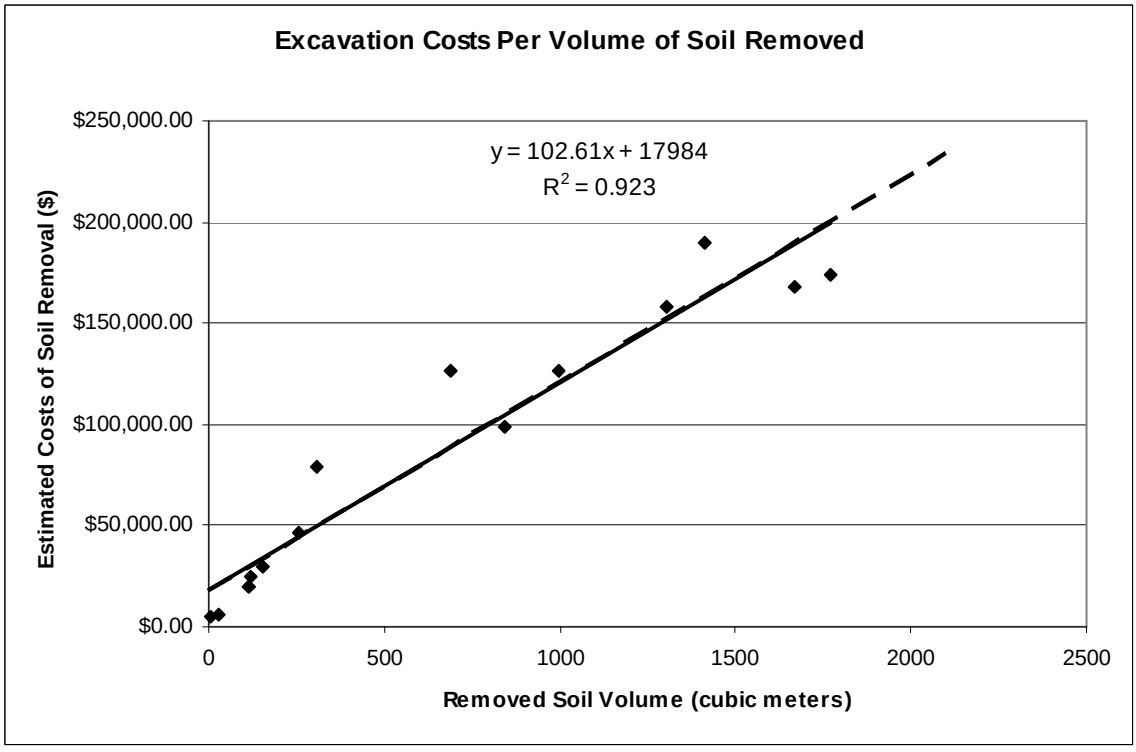


FIGURE 4-4: CORRELATION OF EXCAVATION COSTS TO VOLUME OF SOIL REMOVED

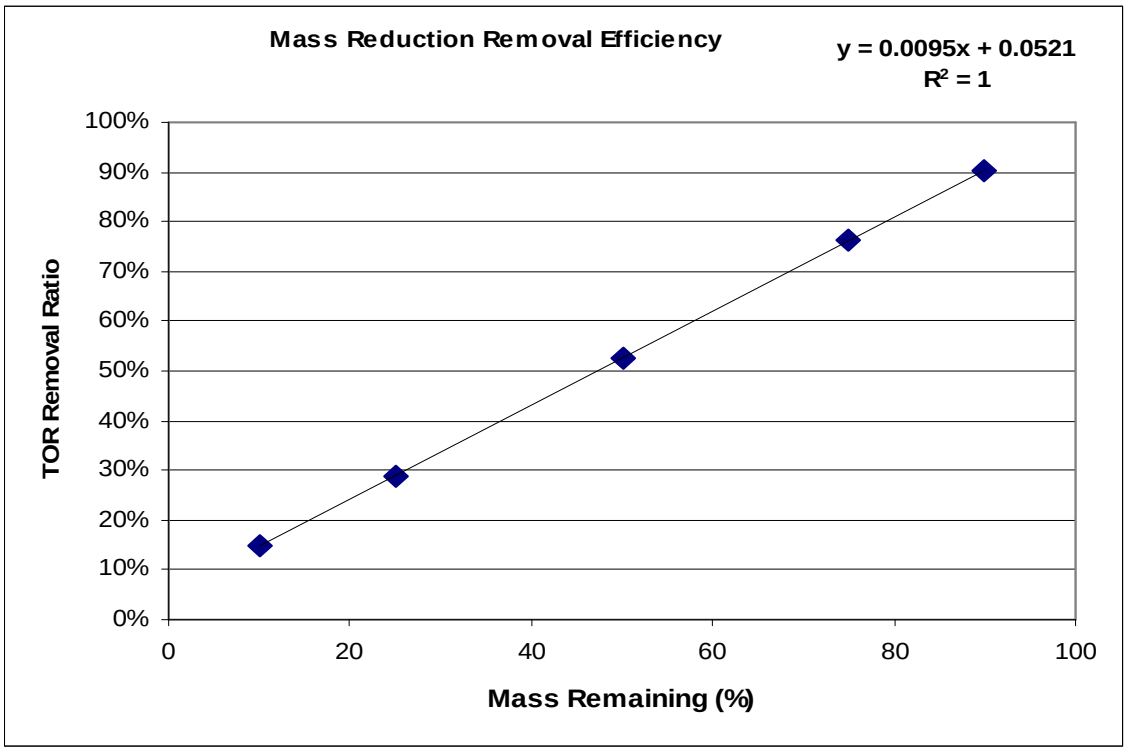


FIGURE 4-4: CORRELATION OF EXCAVATION COSTS TO VOLUME OF SOIL REMOVED

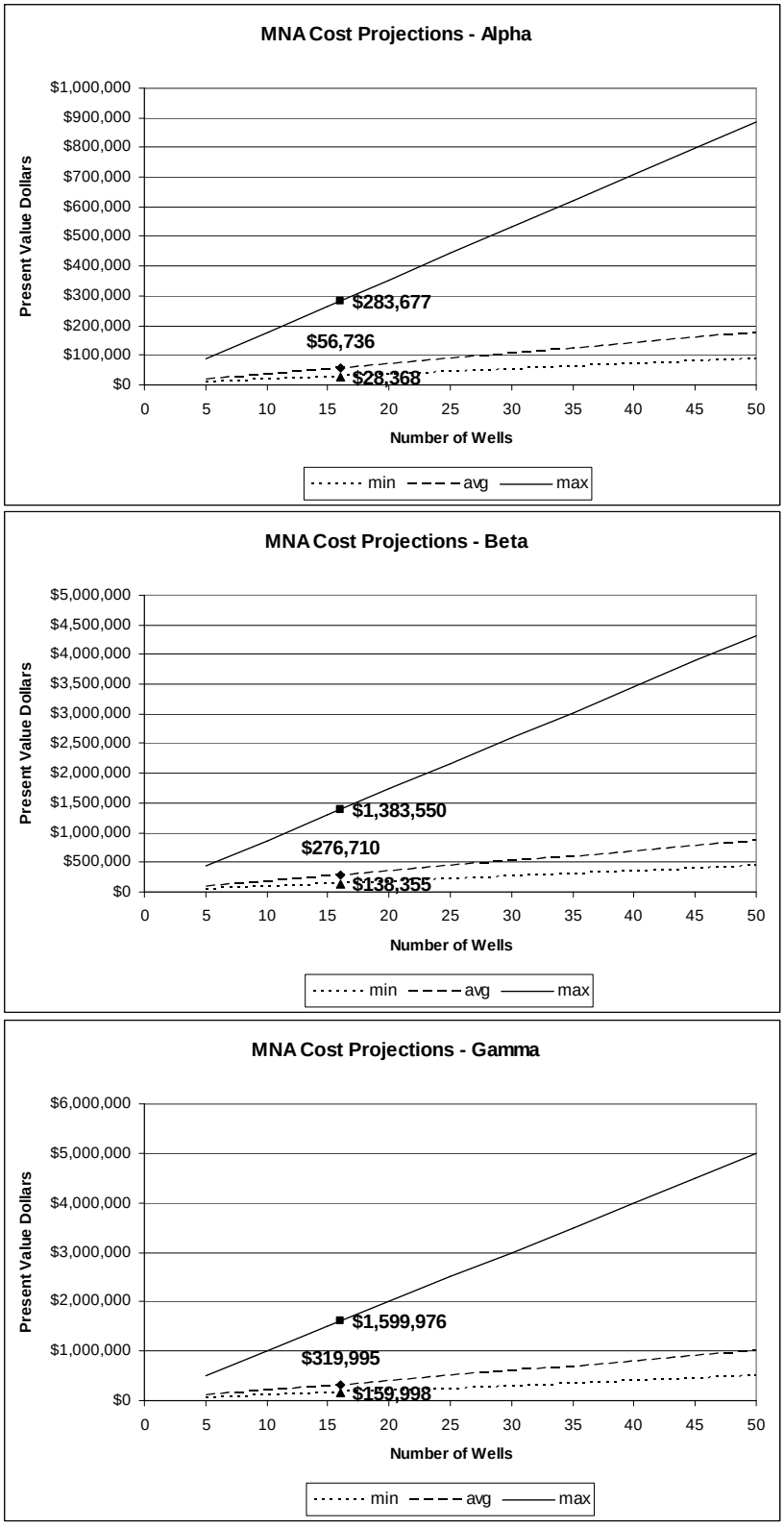
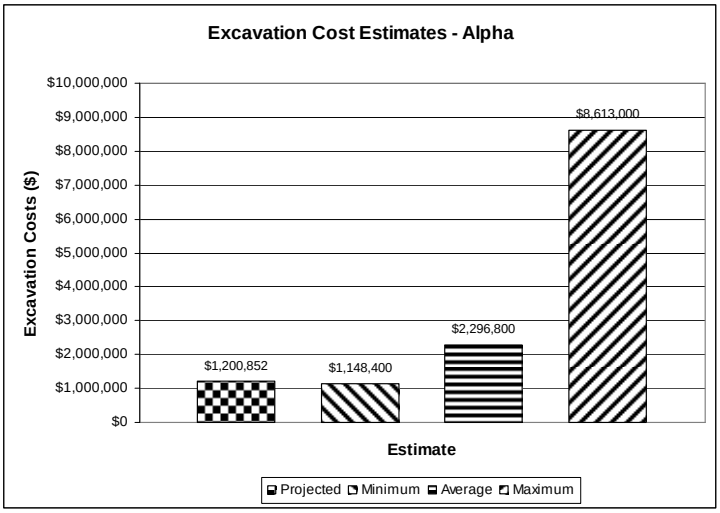
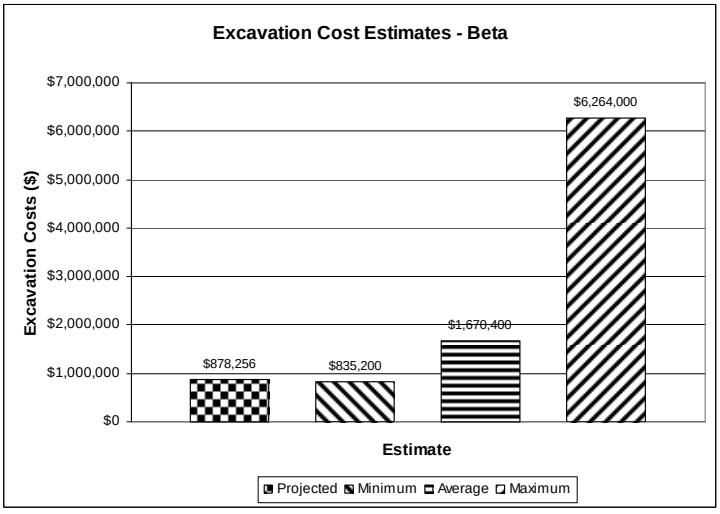


FIGURE 4-5: OPTIMAL MNA NET PRESENT VALUE COST PROJECTIONS FOR LOW TOR (ALPHA), MODERATE TOR (BETA), AND HIGH TOR (GAMMA)





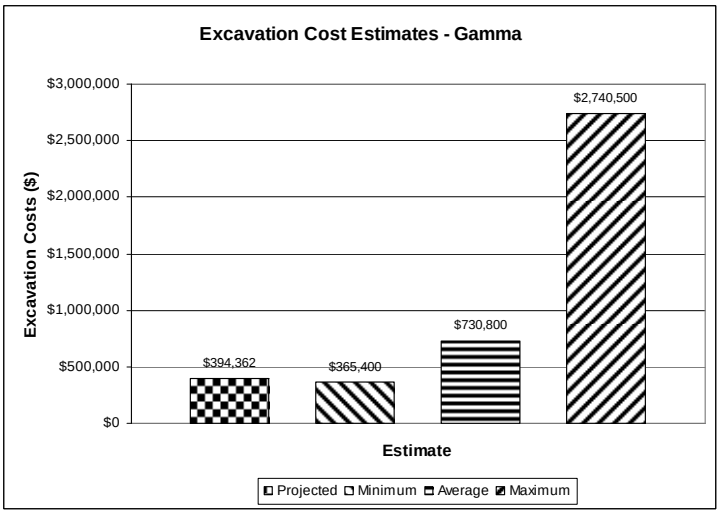
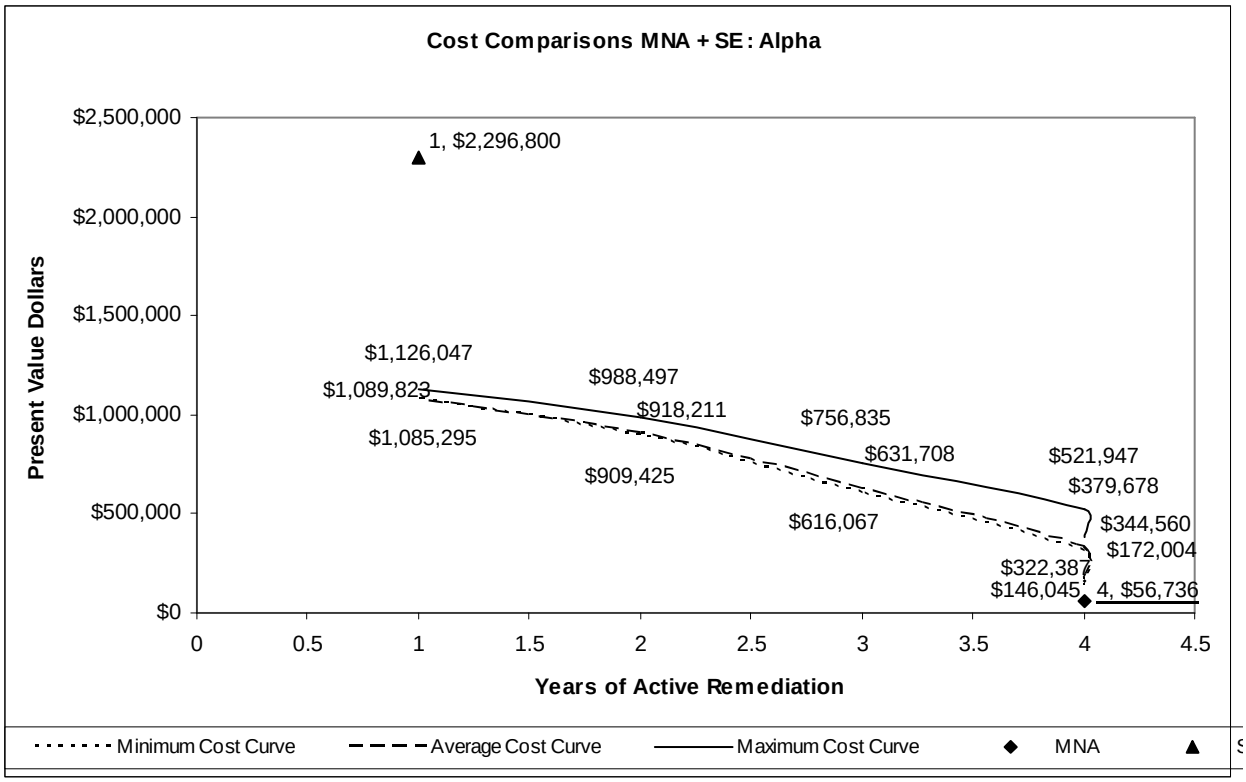


FIGURE 4-6: OPTIMAL SE NET PRESENT VALUE COST PROJECTIONS FOR LOW TOR (ALPHA), MODERATE TOR (BETA), AND HIGH TOR (GAMMA)



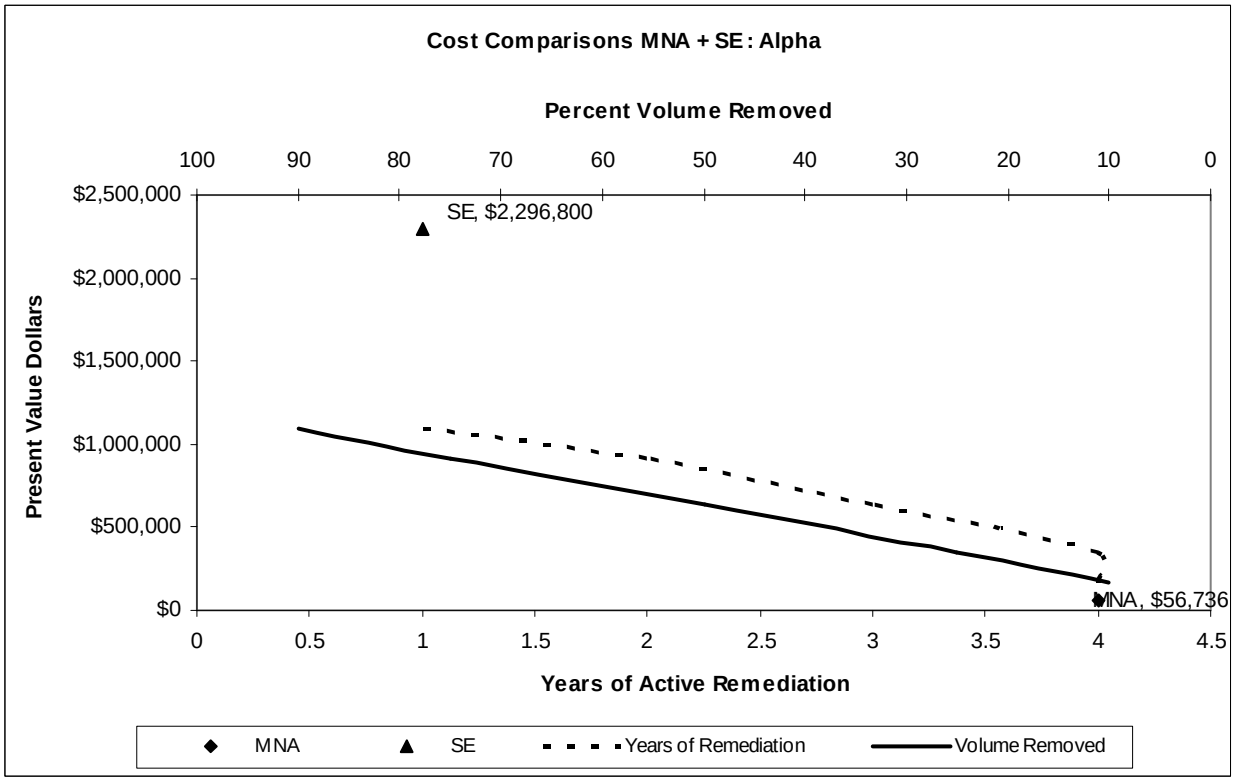
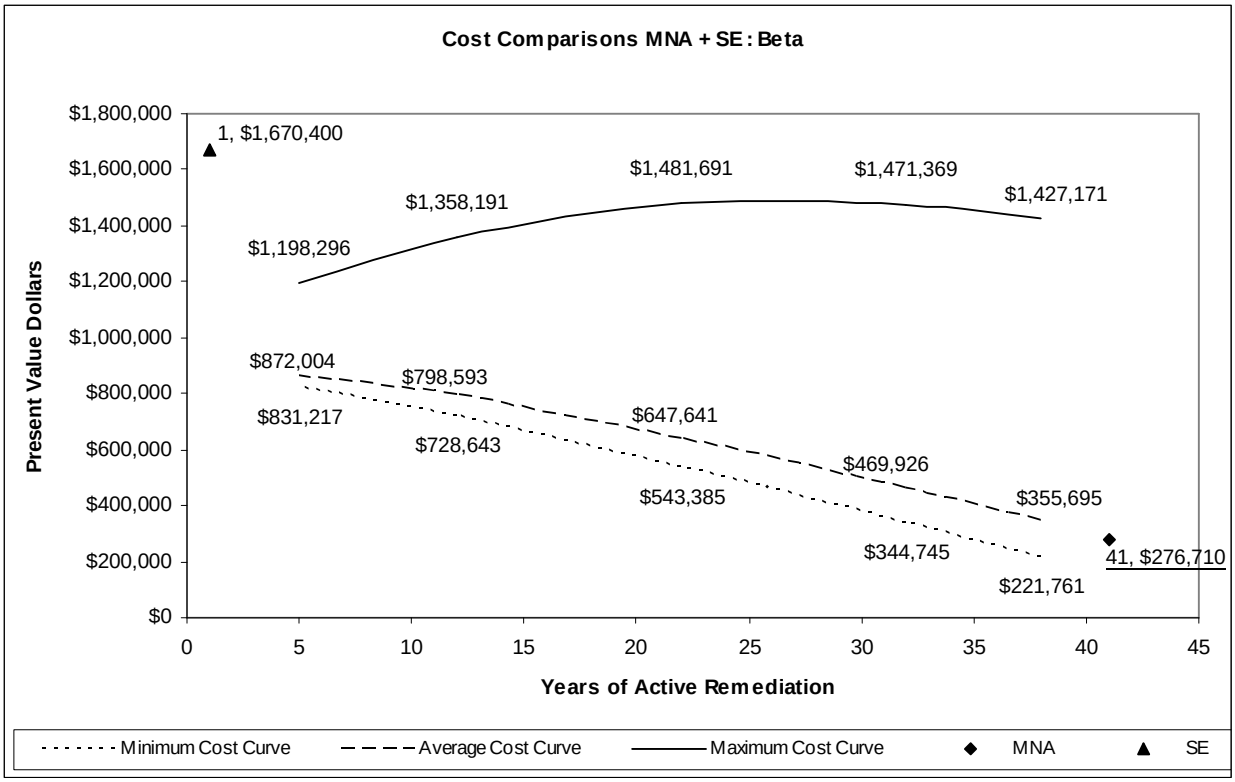


FIGURE 4-7: OPTIMAL SE + MNA NET PRESENT VALUE COST PROJECTIONS – LOW TOR



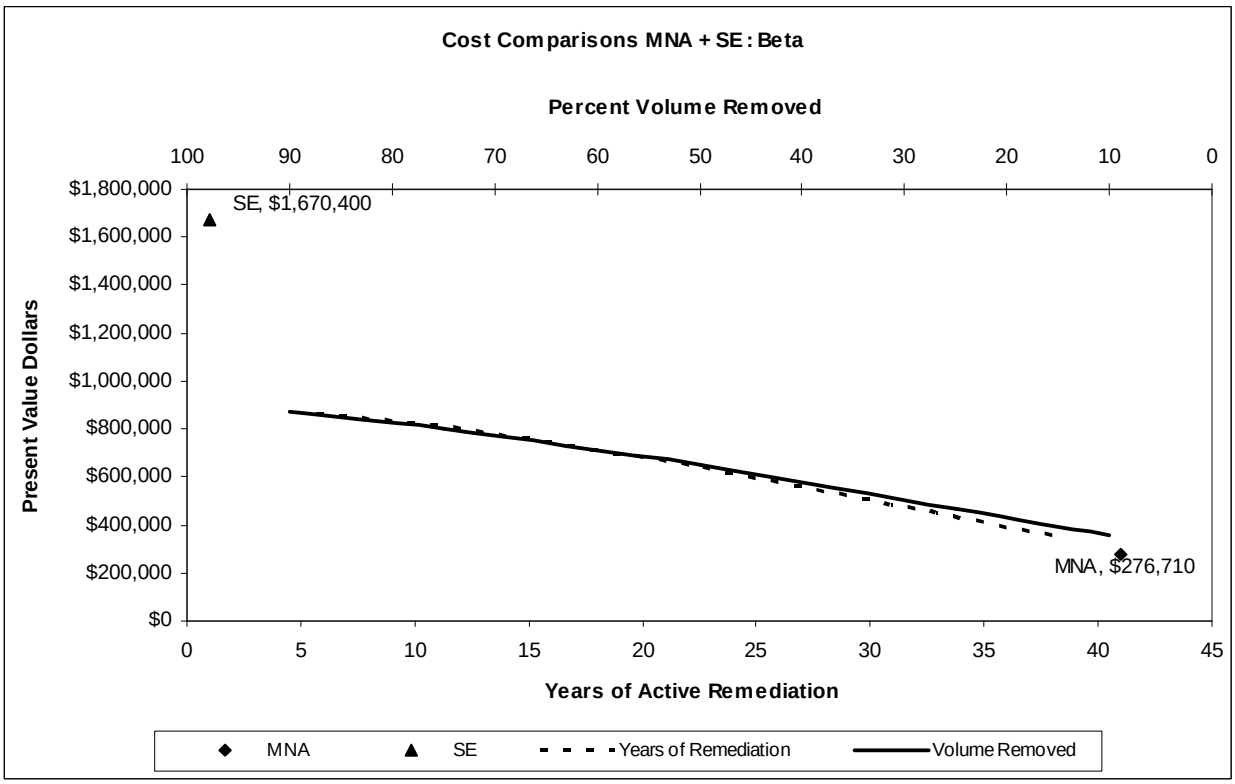
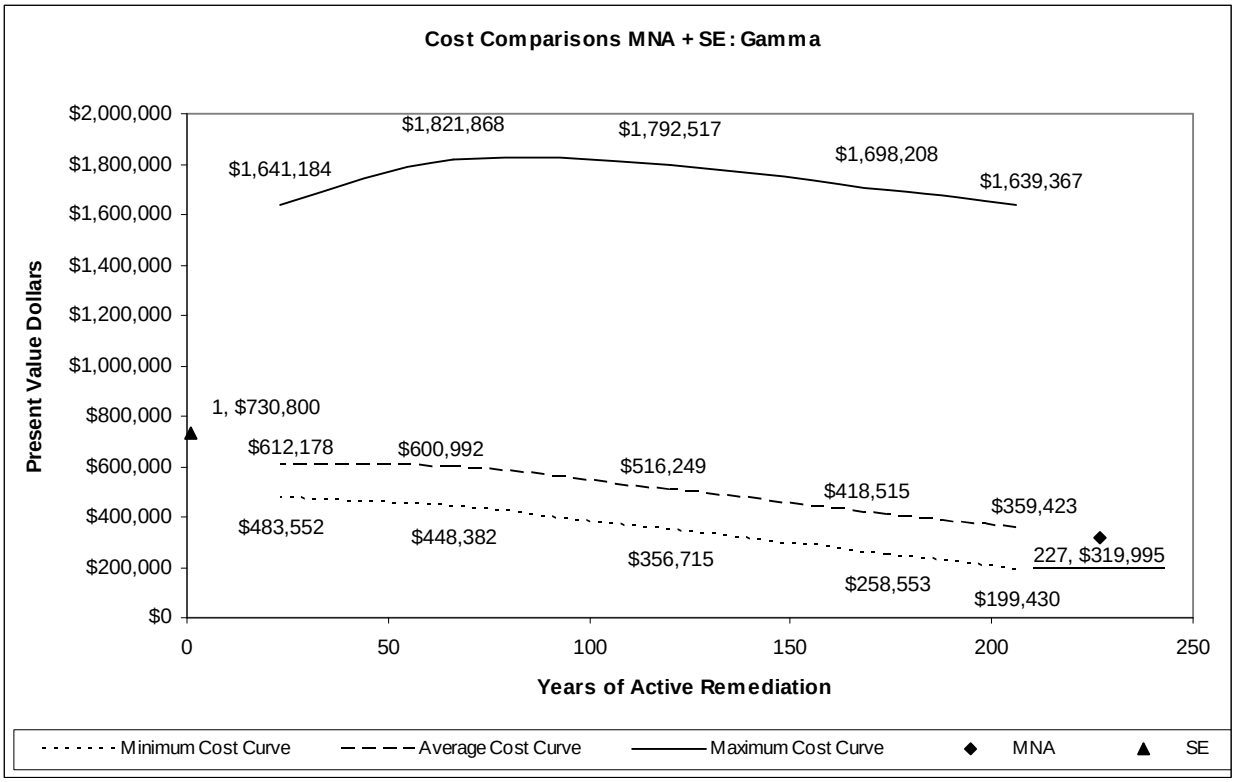


FIGURE 4-8: OPTIMAL SE+MNA NET PRESENT VALUE COST PROJECTIONS–MODERATE TOR



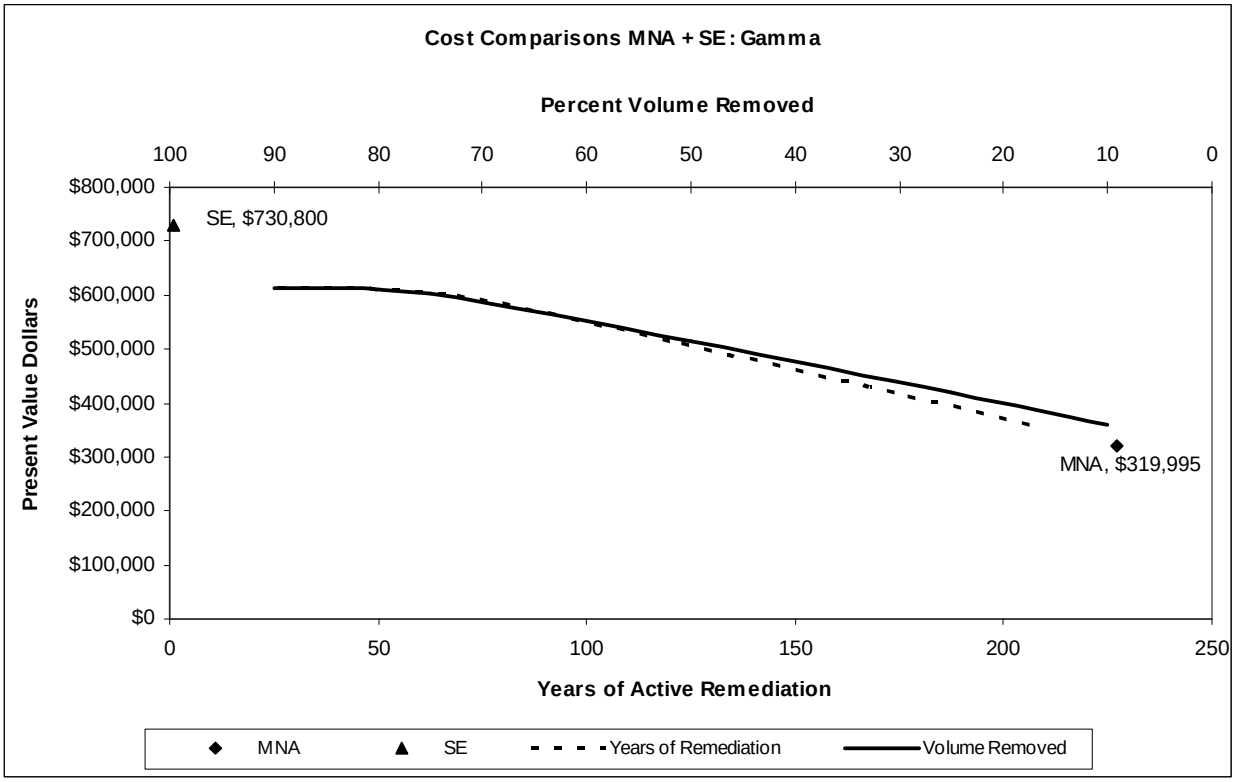


FIGURE 4-9: OPTIMAL SE + MNA NET PRESENT VALUE COST PROJECTIONS – HIGH TOR

CHAPTER FIVE — ASSESSMENT

SCIENCE ▪ ENGINEERING ▪ ECONOMICS ▪ POLICY

5.1 AN EVOLUTIONARY APPROACH TO SITE REMEDIATION

The test of any engineering product is the value added of its application to society. Similarly, the test of a policy application is in its implementation. As engineering is often the bridge from the abstract to the applied, so can public policy be viewed as a bridge from a public need to a public remedy. The final phase of this interdisciplinary exercise underscores the value of an amalgamated approach for advancing the technical solutions of environmental remediation within the highly charged policy arena. The engineering and economics are synthesized into the **S▪E▪E▪P▪AGE** paradigm as illustrated in Figure 5-1. This paradigm is founded upon *sound science for an environmental engineering, effectual economics, and public policy agenda*, which might also be referred to as simply **S▫E▫E▫P** – ***S****cience* ***E****ngineering* ***E****conomics* ***P****olicy*.

As we learn more about the contaminants that pollute our environment, their pathways to receptors of interests, and their associated effects upon those receptors we gain an appreciation for the difficulty that lies in treating thousands contaminants at thousands of sites within differing hydro-geochemical environs. Environmental policy decisions have evolved over the past decade to highlight the case-sensitive nature of the contaminant transport problem. Increasingly, federal and state environmental regulations allow for the specific address of environmental contamination issues through chemi- and site-specific action plans that consider risk as fundamental to the science, engineering, economics, and policy that emerge.

Louisiana is one of many states that has deviated significantly from the *command and control* philosophies of an earlier environmental age and graduated to a pragmatic system of risk-based regulatory mechanisms. During the late 1990s and throughout this decade there has been considerable growth in the development and adaptation of *new* environmental policy instruments (NEPIs) (Jordan et al., 2003). While countries such as Australia, Austria, Finland, France, Germany, Ireland, the Netherlands, and the United Kingdom have seen widespread growth in the use of these policy tools, the United States has begun to see a quantitative increase in the use of NEPIs at the federal, state, and sub-state levels.

The achievements within environmental remediation have been largely limited to successes from two separate and distinct components on opposite ends of the spectrum. Through extensive research and development efforts, environmental technologies have become more efficient and effective at mitigating the impact of contaminants. Furthermore, policy makers have increasingly sought ways to integrate more, and traditionally under-represented, stakeholders into the regulatory process earlier, often, and continuously. Unfortunately, the barriers that have thwarted the translation of sound science and environmental engineering into a practical and politically-expedient public policy agenda are real and costly. While the barriers are often economic in nature, they are sometimes social, political, or technical. Regardless of the nature of the barriers there is an associated cost – a burden that zaps efficiency, effectiveness, implementation, and reward from the system.

Historically, the advance of NEPIs has been the purview of environmental social scientists. Environmental engineers have long avoided the complicity that accompanies the promulgation of environmental laws, mandates and associated regulations. The links that connect and *interconnect* the worlds of policy cannot be ignored. The question, *does science drive policy or policy drive science*, becomes less relevant in the new state of the environment, as both shape and are shaped by the other.

Furthermore, it is impossible to discuss a scientific, engineering or public policy agenda without consideration of the overwhelming influence of economics. A considerable portion of the environmental market works external to the free market mechanics that have passed through and from the great economic philosophers, such as Adam Smith. As a result most environmental projects will require a significantly unique approach to address the fiscal deviations that are characteristic of a public good, and thus subject to policy address.

5.2 The Advance of Voluntary State Remediation Programs

5.2.1 VOLUNTARY EFFORTS IN THE STATE OF LOUISIANA

Site contamination poses a danger to human health and the environment. However, the presence of contaminants may also stand as a barrier for economic growth and redevelopment. The

tenets of Superfund are unforgiving with regards to contaminant liability and mitigation. Often current ownership equates to remedial responsibility even in situations whereby the current owners did not contribute to the actual state of pollutancy. To help combat the hesitancy of business entities from acquiring, developing, managing and maintaining these brownfields-type sites the State of Louisiana passed the Voluntary Investigation and Remedial Action in 1995 to allow "property owners and other persons who clean up properties to risk-based standards to get a Certificate of Completion from the Louisiana Department of Environmental Quality (LDEQ) [releasing] the property owner and any subsequent owners of the property from further liability under state law for the past contamination at the site." This statute, Act 1092, together with the Louisiana Voluntary Remediation Regulations, formalize the Voluntary Remediation Program and help to usher in a new era of voluntary cleanups.

The Risk Evaluation/Corrective Action Program (RECAP) is the primary statutory mandate that pertains to remedial actions in the state of Louisiana. It is designed to "address risks to human health and the environment posed by the release of chemical constituents to the environment," through these criteria used to evaluate risk: (1) determination "if corrective action is necessary for the protection of human health and the environment" and (2) identification of "constituent levels in impacted media that do not pose unacceptable risks to human health or the environment." (LDEQ, 2003) Through a tiered framework including screening and progressive levels of site management, characterization and remediation are tailored specifically to an individual site via appropriately defined site-specific risks (LDEQ, 2003).

LDEQ has stated that its chief reason for establishing RECAP *is the necessity to ensure that risks are properly evaluated to protect human health and the environment*, [because] *absent the establishment of such a program, the Department will expend considerably more resources to ensure that risk is evaluated properly, the regulated community will not have a clear understanding of the Department's requirements, and the general public will be uncertain as to the criteria used by the Department for remedial decisions* (LDEQ, 2003). Of further benefit, RECAP helps to establish clear and consistent multiple-media (air, land, and water) guidelines.

While RECAP is consistent with federal guidelines on risk assessment, it is a state-specific policy instrument that exploits opportunities within US EPA's guidance that might be subject to interpretation relative to appropriate risk level, exposure concentration, groundwater use, land use,

points of exposure, and points of compliance. Additionally, while RECAP establishes a basis for the consistent evaluation of risk it also allows for the advance of more stringent standards as assesses on a case-by-case basis.

RECAP has significantly reduced transaction costs for the regulated community and also for LDEQ. This program has also enhanced the requirements for documents submission while also improving upon the checks and balances for ensuring all documents and all information necessary for remedial decision making are appropriately in place and in service. RECAP has been one contributor to a robust, publicly-accessible database of sites and their associated data, documents, and records of decision – LDEQ's Electronic Document Management System (EDMS) which contains all official records that have been created or received by LDEQ. The information in this database helped to establish a significant portion of the inputs for the enviro-economics model that was developed in Chapter 4 of this document.

5.2.2 VOLUNTARY EFFORTS IN THE STATE OF WYOMING

In looking for a test case, it was observed that a similar development has occurred in the state of Wyoming. In 2000, Wyoming established the Voluntary Remediation Program (VRP) as a response to its legislative body promulgating the Voluntary Remediation of Contaminated Sites (VRCS) law. This program establishes "a process that can be used by owners of contaminated sites, or by potential developers, to quickly reach decisions about required remedial activities and put contaminated sites back into productive reuses." (WDEQ, 2008) The VRP has facilitated the development and implementation of a clearinghouse of materials and information that have been generated as a result of the Wyoming Department of Environmental Quality (WDEQ) producing the necessary guidance documents, policies, and outreach materials that are required for individuals, businesses, and government entities to be able to perform site evaluations and contaminant remediation leading to a "release from future environmental liability." (WDEQ, 2008)

The WDEQ has entered into Memoranda of Understanding (WDEQ, 2002a; WDEQ, 2002b) with the US EPA and its Region VIII relative to the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), also known as Superfund, and the Resource Conservation and Recovery Act (RCRA). These agreements are implementation driven and provide

specific guidance for owners and operators subject to remedial requirements that may include corrective actions, closure, and post-closure requirements. Both entities recognize the need that any activities, including non-actions, be protective of public health and the environment. Further, the cooperating agencies are cognizant of the synergy that is produced when two complementary organizations are working to utilize resources and govern activities in a manner that does not duplicate resources, effort, or activity.

The rulemaking process included extensive and expanded stakeholder input from groups including business and industry, the public and public interests groups, and state and federal regulators. The cleanup objectives are clearly stated through the memoranda (WDEQ, 2002a; WDEQ, 2002b):

1. Remedies are protective of human health and the environment.
2. Facility-wide characterizations are required to determine the full nature and extent of releases.
3. All releases (both on-site and off-site) of hazardous wastes or constituents into the environment are addressed.
4. Meaningful opportunities for public involvement are provided for facilities subject to cleanup.

Furthermore, the adequacy of the VRCS program is mutually defined because of the following characteristics (WDEQ, 2002a; WDEQ, 2002b):

1. Provides opportunities for meaningful community involvement.
2. Ensures that voluntary cleanup actions are protective of human health and the environment.
3. Has adequate resources to ensure that voluntary cleanup actions are conducted in an appropriate and timely manner, and that both technical assistance and streamlined procedures, where appropriate, are available from WDEQ.
4. Provides mechanisms for the written approval of remedy agreements and a certification indicating that the remedy is complete.
5. Provides adequate oversight to ensure that voluntary cleanup actions are conducted in such a manner to assure protection of human health and the environment.
6. Shows the capability, through enforcement or other authorities, of ensuring completion of response actions if the volunteering parties conducting the response actions fail or refuse to

complete the necessary response actions, including operation and maintenance, institutional controls or long term monitoring activities.

WDEQ and US EPA believe that this type of voluntary program provides an added bonus in allowing affected communities to reap the rewards of site remediation through the potential of post-closure site redevelopment.

5.3 CASE STUDY – VOLUNTARY REMEDIATION PROGRAM SITE – CAMPBELL COUNTY, WYOMING

5.3.1 SITE BACKGROUND

The site that is being studied is located in Campbell County, Wyoming. The site is a former natural gas compressor station located approximately two miles north of Gillette, Wyoming. The site was accepted into the WDEQ VRP in 2001.

5.3.2 COAL PRODUCTION IN THE POWDER RIVER BASIN

The state of Wyoming produces more coal than any other state in America. Furthermore, Wyoming's Powder River Basin is responsible for 20 percent of the nation's coal production (PWCC, 2008a). This quantity is due in large part to the demand for this more environmentally friendly coal that has a relatively high level of thermal energy as measured by the British Thermal Unit (BTU), and low sulfur and ash contents. Since the passage of the Clean Air Act (1970) the nation's domestic coal use has risen more than 80 percent; but fortunately there is evidence of decreases among the six nationally regulated pollutants – smog, lead, carbon monoxide, particulates, nitrogen dioxide, and sulfur dioxide (PWCC, 2008b).

There are two major coal fields within the United States known as *Western Coal* or *Eastern Coal*. The coal of Wyoming is associated with *Western Coal*, and while the BTU content is lower than that of *Eastern Coal*, the environmental benefits are a significant factor in the mining of the coal from the western coal fields (University of Wyoming, 2002a). Also, the coal of Wyoming is mostly sub bituminous and contains a minimal amount of bituminous, no anthracites, and no lignite. The lower

BTU potential is balanced to some degree by the great thickness of the coal (University of Wyoming, 2002b).

5.3.3 GEOMORPHOLOGY OF THE POWDER RIVER BASIN

In the state of Wyoming, coal is beneath 54 percent of the state's surface area – representing approximately 53,000 square miles (University of Wyoming, 2002c). A significant number of the coal mines within the state are strip mines due to the close proximity of the deposits to the surface. One hundred percent of the state's coal was formed from the accumulation of peat in subtropical swamps from 100,000,000 to 45,000,000 years ago, coinciding with the Upper Cretaceous, Paleocene, and Eocence epochs. The thickness of the coal deposits in the Powder River Basin which are generally 50 to 100 feet thick, but have been measured to reach almost 300 feet provide a clear picture of the geomorphology (University of Wyoming, 2002d; 2002e):

- The formations began in a freshwater wetland because thick coals originate from peat that is produced by land-based woody plants.
- The wetland was a swamp with trees as opposed to a grassy marsh or a mossy bog as evidenced by microscopic tissue analysis.
- Stagnant water was present in the wetlands because the degree of decay would have been inconsistent with a moving (oxygenated) water, the pH due to decay had to remain acidic, thus limiting decay and preserving much of the peat, and flowing water would have carried away a significant amount of the gathering peat.
- The peat had to remain submerged in stagnant and acidic water to stave off complete oxidation and loss.
- The subsidence of the basin was continuous or the rise of the water table was slow enough to produce the necessary space to hold the growing peat deposits.
- The rate of swamp subsidence was balanced appropriately to prevent the formation of a lake or conversion to dry land.
- The optimal conditions for the growth of peat deposits and the eventual formation of coal had to occur at a relative constant over geologic time as approximately 1 millimeter of peat accumulates in a one-year time period.

- The ideal climate for peat formation in Wyoming averaged about 80 degrees Farenheit with a yearly rainfall of 120 inches.

The mountains of Wyoming would rise in cycles that would cover the swamps with sediment and then see the swamps reform for thousands of years until the cycle would repeat itself hundreds of times over. The swamps would disappear and the basin would fill with thousands of feet of rock covering the peat as the climate cooled and dried. Coal formation would coincide with the rising temperature of the peat. The erosion of this basin has been a major influence in the evolution of this coal bed and its ideal characteristics for open pit, or strip mining as the coal is very near the ground surface (University of Wyoming, 2002f).

5.3.4 ENVIRONMENTAL CONDITIONS WITHIN THE POWDER RIVER BASIN

5.3.4.1 WATER RESOURCES

Within the PRB, surface and groundwater are used to supply agriculture (surface), municipal (surface and groundwater), and domestic and industrial (groundwater) water resource needs. The Big Horn Mountains supply most of the surface water runoff to the basin's primary waterways, the Powder and Tongue Rivers, which has fostered substantial agricultural development along the drainage pathways. Reservoirs are also used extensively within the basin for agriculture and municipal demand (Bureau of Land Management, 2006).

According to January, 2002 statistics, "approximately 161,160 acres of land were actively irrigated in the Powder/Tongue River Bain," which results in about 194,000, 184,000, and 178,000 acre-feet of water depletion per year during wet, normal, and dry years, respectively. The amount of groundwater used for irrigation was 300, 200, and 200 acre-feet for the same wet, normal and dry years, respectively (Bureau of Land Management, 2006).

Twenty public water supply systems (incorporated municipalities, water districts, and privately owned water systems) supply all but two communities within the basin with water from within the basin. Four systems are supplied with 2,700 acre-feet of surface water per year, while the other 16 systems use 500 acre-feet per year of groundwater. Annual domestic demands that range

between 2,400 and 4,400 acre-feet are supplied by groundwater sources (Bureau of Land Management, 2006).

The major industrial water needs within the PRB arise from conventional oil and gas production, along with coal bed natural gas development. Together, these activities require approximately 68,000 acre-feet per year and is supplied by groundwater. Recreational and environmental uses are deemed non-consumptive and are thus not quantified. However, a major source of water loss is through evaporation in 14 key storage reservoirs within the basin. This loss is approximately 11,300 acre-feet of water (surface) per year. In total, approximately 281,200 (surface: 208,000; ground: 73,200), 271,100 (198,000; 73,100), and 265,100 (192,000; 73,100) acre-feet of water are used within the PRB in any given year (Bureau of Land Management, 2006).

Most of the natural flow of surface water to the PRB is supplied by five rivers or streams: Little Bighorn River, Tongue River, Powder River, Crazy Woman Creek, and Piney Creek. Irrigation practices also influence many of the other major drainages. Wyoming is also required to share some of the physically available surface water in the basin with the state of Montana, via the Yellowstone River Compact (Bureau of Land Management, 2006).

The major aquifers of the PRB which are available for water supply are the Madison, Datoka, Fox Hills/Lance, Fort Union/Wasatch, and Quaternary Alluvial Aquifer Systems. The deepest is the Madison Aquifer which lies within Paleozioic Tensleep Sandstone, Amsden Formation, Madison Limestone, Bighorn Dolomite, and Flathead Sandstone. The thickest unit is the Madison Limestone ranging from 200 to 1,100 feet in thickness and it is the deepest and the most vital high-yield aquifer for the state of Wyoming. The range in transmissivity is from 500 to 90,000 gallons day per foot (gpd/ft) with well yields up to 4,000 gallons per minute (gpm). The quality of water from the Madison Limestone is very good; there is mainly calcium-magnesium bicarbonate with high pockets of fluoride and radionuclides. The range for total dissolved solids (TDS) is from 600 to 3,000 milligrams per liter (mg/L). The higher measures are primarily the result of sulfates and chlorides. Within the PRB, depths to the Madison Limestone range from about 6,000 feet (east of Gillette, Wyoming) to almost 16,000 feet (SW PRB). The average annual recharge to the Madison Limestone is about 75,000 acre-feet (Bureau of Land Management, 2006).

The Dakota Aquifer System is comprised to two major formations: the Cloverly Formation and the Newcastle Sandstone. Their total thickness is approximately 200 feet and their yields range from a low of 1 to 40 gpm to a high of 250 gpm. Transmissivity ranges from 7 to 230 gpd/ft. Sodium bicarbonate is predominant in the Dakota and TDS range from 300 to 3,000 mg/L. Low well yields inhibit the Dakota from serving as a major water supply (Bureau of Land Management, 2006).

The Fox Hills/Lance Aquifer System is comprised of the Lance Formation and the underlying Fox Hills Sandstone. The thickness of the Lance Formation spans 600 to 3,000 feet and yields no more than 15 gpm. Transmissivity ranges from 76 to 2,100 gpd/ft. The water quality within the Lance Formation is heavily influenced by sodium sulfate or calcium sulfate and TDS range up to 3,000 mg/L. The presence of sodium prohibits the this water being used for irrigation, stock or domestic purposes (Bureau of Land Management, 2006). The thickness of the Fox Hills Sandstone may reach 700 feet and yield approximately 15 to 50 gpm. Transmissivity ranges from 76 to 1,600 gpd/ft. The municipal water supply for Gillette, Wyoming utilizes wells within the Fox Hill formation and draws 85 to 705 gpm. The water quality resembles that of the Lance and does limit its application to domestic or stock uses. Fluoride can also be limiting in local hot spots. The depth to formation is approximately 1,000 feet in the majority of the PRB (Bureau of Land Management, 2006).

The Fort Union/Wasatch Aquifer System is made up of a local aquifer formation – Wasatch – and a regional aquifer formation – Fort Union. The thickness of the Wasatch spans 500 to 2,000 feet while the Fort Union configuration spans 1,200 to 3,900 feet in thickness. The Wasatch is composed of "fine to coarse-grained lenticular sandstone with interbedded shale and coal" while the Fort Union is a "fine- to medium-grained siltstone with abundant coal and shale." The transmissivity of the Wasatch and Fort Union formations ranges from 520 to 2,200 gpd/ft and 10 to 95 gpd/ft, respectively. Well yields are commonly less than 15 gpm for the Wasatch and for the Fort Union, low yields range from 1 to 60 gpm, while 250 gpm productivity is possible. The TDS within the Wasatch measure 141 to 6,620 mg/L, sulfate content can reach 4,000 mg/L and iron values may reach 25 mg/L. Within the Fort Union formation, these conentrations range from 484 to 4,630 mg/L (TDS), less than or equal to 1,870 mg/L (sulfate), and approaching 19 mg/L (iron). Both the Wasatch and Fort Union formations are used locally to supply domestic and stock needs.

The Wasatch is not considered acceptable for irrigation due to elevated sodium levels. The Fort Union formation is used extensively to pump from CBNG wells at average rates between 12 and 45 gpm (Bureau of Land Management, 2006).

The Quaternary Alluvial Aquifer System is a locally natural development located in alluvium and terrace deposits near the major drainages within the PRB. The thickness of the alluvium is generally less than 100 feet. In some areas, well yields may reach 50 to 300 gpm. The upper limit for transmissivity is approximately 20,300 gpd/ft. The chemistry of the water is highly variable across the formation as TDS values may reach 4,000 mg/L. This water is a source for municipal, domestic, and stock purposes. The stream-aquifer interaction is significant as a hydraulic process. These alluvial aquifers are very shallow and are recharged by either groundwater from the Wasatch Aquifer below or stream infiltration (Bureau of Land Management, 2006).

5.3.4.2 Social and Economic Conditions

The primary influence upon the socioeconomic station of the Powder River Basin of Wyoming has been the energy development that has transpired for well over 100 years. The first coal mine became operational in 1883. It was in the 1970s and 1980s that the PRB became the stalwart coal producing region that now leads the country in coal production. For over 50 years now, the PRB has experienced continuous economic growth and development. These economic benefits have translated to an increased stresses within communities relative to social structures, housing resources, and public infrastructure and service systems. However the benefits have also fostered an increased capacity to generate positive responses to the aforementioned conditions (Bureau of Land Management, 2005d).

If population growth can be used as a metric for cumulative social and economic change within the PRB, it suggests that the cycles of energy development are highly correlative with the economic and social booms that have seen the population of Campbell County increase more than 300 percent from 1920 to 1980. Furthermore through the start of the 21st century, Campbell County has experienced the highest rate of growth of all Wyoming counties which can be directly linked to coal bed natural gas (CBNG) development (Bureau of Land Management, 2005d).

5.3.5 THE REMEDIAL IMPERATIVE

There have been a number of studies relating the natural attenuation capacity of various hydrogeologic media to the fate and transport of non-aqueous phase liquids. However, there has been limited documentation of the natural attenuation capacity of coal aquifers. One of the factors that peak research interests center around the high carbon content of sub-bituminous coal, potentially increasing the sorptive capacity of such a coal aquifer. Also interesting within the research are the roles of microorganisms and dissolved oxygen concentrations that may support or limit the natural attenuation capacity of the aquifer.

From Widdowson et al. (personal communication), the following is observed

> *While it is clear from this work that contaminant transport is being strongly retarded by both sorption and biodegradation, it is also evident from the conceptual model and several quarters of groundwater quality sampling that contaminant concentrations are not declining, indicating that a fairly large area of the coal aquifer, where it has been most heavily impacted, is a continuing source of groundwater contamination that will need to be addressed before progress toward WDEQ cleanup standards can be achieved. Based on the lack of free product at monitoring wells installed in the most contaminated areas of the coal aquifer, relative to dissolved petroleum hydrocarbon concentrations, it appears that high concentrations of natural gas condensate sorbed by coal aquifer matrix, primarily in identified source areas, is the primary continuing source of groundwater contamination.*

Thus the coal aquifer located below the site provides an ideal case for exploring the economic implications of monitored natural attenuation (MNA) as a singular or joint remediation strategy.

Investigations by Widdowson et al. reveal three likely porosity characterizations along with three probable velocity and decay rate scenarios as described in Table 5-1. The TOR formulation developed previously is used to determine the TOR associated with MNA based on benzene concentrations (Equations 3-35):

$$TOR_{Benzene} = -6.052 \cdot Sr \cdot \ln(T) \cdot RCC + 1576.571 \cdot Sr \cdot \ln(T) + 37.841 \cdot Sr \cdot RCC - 0.006 \cdot \ln(T) \cdot RCC - 9849.268 \cdot Sr + 2.929 \cdot \ln(T) + 0.036 \cdot RCC - 18.333$$

where T = dimensionless travel time,

RCC = Regulatory Compliance Concentration, and

Sr = residual saturation.

The dimensionless travel time variable is a function of source length, groundwater velocity, longitudinal dispersivity, and decay rate.

The remaining inputs that are required for the socio-economic model are summarized in Table 5-2 and include residual saturation; length, width, thickness, and bulk density for both the source and plume; depth to source; interest rate; and the minimum, average, and maximum costs per well. As shown in Chapter 4, net present value is determined using Equation 4-1:

$$P = A\frac{(1+i)^n - 1}{i(1+i)^n}$$

Here, P represents the present value of an annuity or annual expense (A) evaluated over a number of years (n) at an associated rate or interest (i). The annuity is the product of the number of wells and the annual cost associated with each well and defined according to via Equation 4-2:

$$A = W_{\$} * N_W$$

5.3.6 EVALUATION OF REMEDIATION ALTERNATIVES

The above scenarios were processed through the economic model developed in the previous chapter. With the goal of determining the benefits of a single technology approach to site remediation of this coal aquifer located within the Powder River Basin versus a multi-technology approach to site cleanup. The results presented in Table 5-3 show the costs to remediate the site using the singular technologies of MNA and source zone excavation (SE).

In evaluating the costs of excavating the source versus the costs for performing MNA, it becomes clear that MNA is a cost-effective strategy. However, it must be applied in situations that have been approved by the appropriate state and federal regulatory entities. These situations usually require that remediation occur in a timely fashion. Unfortunately, MNA cleanup times in excess of 50 – 100 years tend not to be favorably received by regulators, legislators, or the public. According to the economic model MNA would cost between $400,000 and $2,800,000. While these numbers are economically desirable, 100 to 1000 years of site monitoring and data collection are infeasible and impractical.

Site excavation in the majority of cases tends to be a low-effort technology that has the advantage of being a more physically brutish solution. The advantage of an excavation strategy is in its high rate of acceptance as a remedial alternative and in the time savings that decrease the time to site closure. The costs of excavating the coal aquifer source at the site would range between approximately $800,000 and $6,000,000, and average nearly $1,600,000. The tradeoff that is highly desirable is that for source zone remediation, it is assumed that the entire excavation could be accomplished in the time span of one year. This one-year active remediation period is the time associated with the excavation remediation option. The remediation of an actual site via soil excavation can never achieve 100 percent mass removal. In the majority of cases, some level of site follow-up, monitoring, and reporting will be required. It is not considered financial or temporally with this option because these costs tend to span almost any remediation option.

A closer look at these alternatives reveals insightful details about the time and cost determinations. MNA costs increase as the velocity and decay rate decrease for each of the constant porosity Groups A, B and C. As comparisons across groups are made, it is shown that MNA costs also increase with a relative increase in porosity. As expected, SE costs are constant within and across groupings. To best understand the tradeoffs that result from the economic projections, a subset of alternatives is considered for further discussion.

Consider the years to remediate for each of the Alternatives, A2, B1, and C3 range from the extremes of 1 year (SE) to 98, 5, and 1,176 years (MNA), respectively. The costs inversely range from $1,536,000 (SE) to $1,142,000, $330,000, and $1,143,000 (MNA), respectively. Figures 5-2, 5-3, and 5-4 illustrate these cost projections more fully along with the cost and remediation curves for the joint technology application of SE + MNA. As expected the costs for SE only would more closely match the joint technology application that removes the most volume of contaminated soil and then employs MNA. Conversely, the costs for MNA most closely pair with the combined technology approach that removes the least volume of contaminated media. The benefit of this modeled approach is in the ability to compare across technology applications and to compare the degree of pairing that might be evaluated. It is critical to point out that an extremely high or low estimate of annual well costs could have a disastrous effect upon the model results. If care is taken to specific the most realistic of estimates, there will be value added.

It can be generalized from the first of the two respective charts for each alternative that for Alternative A2 the spread between SE, MNA, and SE + MNA is relatively small. Alternative B2 shows the case in which SE is significantly higher than both MNA and the average cost of the jointly applied SE + MNA. Lastly, Alternative C3 presents the situation that shows the costs of SE relatively equivalent to the cost of MNA, but less than a significant portion of the average cost curve for SE + MNA. Given this information, it is important to look at the costs in the context of the time to cleanup, which are given by the second of the two respective charts for SE, MNA, and the average cost curve only for SE + MNA. These figures also allow for an analysis of the percentage of contaminated media volume that would need to be removed in correspondence with the years of active remediation.

It is highly unlikely that MNA only would be considered an acceptable strategy for any alternative other than B1 because of the extremely lengthy TORs. Soil Excavation, however might be deemed appropriate as a stand alone technology for either of alternatives A2 or C3, but not for B1, due to a relative doubling of the costs of any other strategy. Finally, SE + MNA is most likely a potential technology candidate for alternatives A2 and B1. It would most likely not be considered for alternative C3 because of both cost and time factors.

In these types of analyses time and cost are major factors in the decision making process. Times of remediation that span to and beyond one hundred years are not highly favored by regulatory agencies or affected citizens. Thus it becomes a matter of practical value to evaluate the options involving a combination of technologies, namely employing a site excavation strategy with an MNA approach. According to the economic results detailed in Tables 5-4 – Tables 5-12, it is clear that a combination of MNA and Site Excavation may be of practical value relative to a remediation regimen. The cost and time considerations are the two most critical factors in selecting a remediation option. Fortunately, state and federal support is often available to defray the full financial burden of cleanup. This aid often spurs the likelihood of site cleanup and increases the chance of an acceptable closure strategy. Furthermore, when a conventional technology is employed with an MNA approach, there is increased acceptance from all stakeholders that fear the appearance of a *do-nothing* stigma and are encouraged by an increased time-to-closure than would be present in an MNA-only alternative.

5.4 CONCLUSION

Science, engineering, economics, nor public policy are new concepts. However, in today's world it has become ever noticeable that there is a correlation that cannot be ignored or minimized. As States become more responsive to the needs and concerns of their citizenry, and adopt new and innovative programs to address past, present and future contamination, the progress observed in states, such as Louisiana and Wyoming, will continue.

Protective of public health standards cannot be created or enforced in a vacuum irrespective of real economic constraints. The remedial options for an individual site are seldom reduced to an obviously singular approach, technology or solution. However, with the proper due diligence a legitimate and effective framework for evaluating real solutions and real consequences can be achieved.

The most obvious solution is to adapt a systems approach toward environmental problem solving that addresses said problems, not from either of separate spectrum ends, but from both ends and both middles simultaneously and synergistically. The S•E•E•P paradigm exhorts a holistic approach toward a unifying concept of engineering environmental policy, heretofore, known as *environmental policy engineering*. Environmental solutions must be built upon sound science that is fostered through repetitive validation of the nature and behavior of our hydro-geochemical complexities. This knowledge must theretofore be applied toward environmental engineering applications that benefit the whole of society. Benefits, however, do not accrue in a vacuum, but are the result of allocations toward an environmentally desirable outcome. In short, benefits cost, and costs must be paid. The greatest of environmental solutions that carry too high a cost are not really solutions at all. Certainly, economics cannot be the only driver of the desirability of an environmental strategy, but it must be a constant and continuous factor in the process. Lastly, sound policy is needed not only because the mechanisms of government demand it but because our citizenry deserve it. The environmental setting is often viewed within a context of litigation and stagnation, thus policy for the people must be sound. Because the atmosphere of environmental remediation contains an element of risk and harm, public policy must be of benefit to society.

The distinction for the S•E•E•P archetype lies in the recognition of multiple linkages and interconnectivities. Science is extended through engineering and through economics into public

policy as policy connects through economics into engineering and back into the fundamentals of science. Furthermore, science influences policy as policy also drives science. The connections between science and economics, as well as policy and engineering are also reciprocal connectors.

5.5 CHAPTER FIVE TABLES

TABLE 5-1: CASE STUDY IDENTIFIERS

Case Identifier	A1	A2	A3
Porosity	2%	2%	2%
Velocity	0.778 m/d	0.286 m/d	0.03 m/d
Decay Rate	0.0044 d^{-1}	0.00161 d^{-1}	0.00017 d^{-1}
Case Identifier	B1	B2	B3
Porosity	3%	3%	3%
Velocity	0.519 m/d	0.191 m/d	0.002 m/d
Decay Rate	0.00293 d^{-1}	0.00108 d^{-1}	0.00011 d^{-1}
Case Identifier	C1	C2	C3
Porosity	5%	5%	5%
Velocity	0.311 m/d	0.114 m/d	0.012 m/d
Decay Rate	0.00176 d^{-1}	0.00065 d^{-1}	0.00007 d^{-1}

TABLE 5-2: CASE STUDY INPUTS

Residual Saturation	0.2	Dispersivity (m)	0.5	Interest Rate	7 %
Source Length (m)	18	Plume Length (m)	150	Minimum Annual Costs per Well	$3,500
Source Width (m)	30	Plume Width (m)	38	Average Annual Costs per Well	$5,000
Source Thickness (m)	1	Plume Thickness (m)	2	Maximum Annual Costs per Well	$12,000
Source Zone Bulk Density (kg/cu. m)	1,600	Plume Bulk Density (kg/cu. m)	1,600	-	-
Depth to Source (m)	10	-	-	-	-

TABLE 5-3: ECONOMIC PROJECTIONS FOR SINGLE TECHNOLOGY REMEDIATION ALTERNATIVES

Case	Number of Wells	Years to Cleanup	MNA Costs ($)			SE Costs ($)		
			Minimum	Average	Maximum	Minimum	Average	Maximum
A1		5	229,611	328,016	787,238			
A2	16	98	798,945	1,141,350	2,739,238	768,000	1,536,000	5,760,000
A3		870	800,000	1,142,858	2,742,858			
						Projected Costs: $809,040		
B1		5	229,611	328,016	787,238			
B2	16	235	800,000	1,142,857	2,742,857	768,000	1,536,000	5,760,000
B3		1,226	800,000	1,142,858	2,742,858			
						Projected Costs: $809,040		
C1		67	791,403	1,130,575	2,713,380			
C2	16	410	800,000	1,142,858	2,742,858	768,000	1,536,000	5,760,000
C3		1,176	800,000	1,142,858	2,742,858			
						Projected Costs: $809,040		

TABLE 5-4: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO A1

Technology	Years to Remediate	Average Costs	Cost Curves		
			Minimum	Average	Maximum
MNA	5	$328,016			
SE	1	$1,536,000			
SE + MNA: 10% Volume Removed	5	-	$292,304	$382,905	$805,706
SE + MNA: 25% Volume Removed	5	-	$384,512	$462,620	$827,122
SE + MNA: 50% Volume Removed	3	-	$535,177	$591,173	$852,486
SE + MNA: 75% Volume Removed	2	-	$681,346	$713,303	$862,433
SE + MNA: 90% Volume Removed	1	-	$766,973	$783,617	$861,289

TABLE 5-5: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO A2

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	98 1	$1,141,350 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	89	-	$878,950	$1,220,970	$2,817,061
SE + MNA: 25% Volume Removed	76	-	$996,982	$1,337,577	$2,927,019
SE + MNA: 50% Volume Removed	52	-	$1,180,241	$1,512,692	$3,064,132
SE + MNA: 75% Volume Removed	29	-	$1,283,655	$1,573,744	$2,927,494
SE + MNA: 90% Volume Removed	10	-	$1,226,489	$1,440,069	$2,436,775

TABLE 5-6: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO A3

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	870 1	$1,142,858 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	790	-	$880,904	$1,223,762	$2,823,762
SE + MNA: 25% Volume Removed	666	-	$1,002,260	$1,345,118	$2,945,118
SE + MNA: 50% Volume Removed	459	-	$1,204,520	$1,547,378	$3,147,378
SE + MNA: 75% Volume Removed	253	-	$1,406,780	$1,749,638	$3,349,637
SE + MNA: 90% Volume Removed	87	-	$1,527,998	$1,870,795	$3,470,517

TABLE 5-7: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO B1

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA	5	$328,016	Minimum	Average	Maximum
SE	1	$1,536,000			
SE + MNA: 10% Volume Removed	5	-	$292,304	$382,905	$805,706
SE + MNA: 25% Volume Removed	5	-	$384,512	$462,620	$827,122
SE + MNA: 50% Volume Removed	3	-	$535,177	$591,173	$852,486
SE + MNA: 75% Volume Removed	2	-	$681,346	$713,303	$862,433
SE + MNA: 90% Volume Removed	1	-	$766,973	$783,617	$861,289

TABLE 5-8: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO B2

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA	235	$1,142,857	Minimum	Average	Maximum
SE	1	$1,536,000			
SE + MNA: 10% Volume Removed	214	-	$880,994	$1,223,761	$2,823,760
SE + MNA: 25% Volume Removed	180	-	$1,002,256	$1,345,110	$2,945,099
SE + MNA: 50% Volume Removed	124	-	$1,204,337	$1,547,116	$3,146,749
SE + MNA: 75% Volume Removed	69	-	$1,396,382	$1,734,783	$3,313,987
SE + MNA: 90% Volume Removed	24	-	$1,450,986	$1,760,779	$3,206,479

TABLE 5-9: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO B3

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	1226 1	$1,142,858 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	1113	-	$880,904	$1,223,762	$2,823,762
SE + MNA: 25% Volume Removed	938	-	$1,002,260	$1,345,118	$2,945,118
SE + MNA: 50% Volume Removed	647	-	$1,204,520	$1,547,378	$3,147,378
SE + MNA: 75% Volume Removed	356	-	$1,406,780	$1,749,638	$3,349,637
SE + MNA: 90% Volume Removed	123	-	$1,528,132	$1,870,988	$3,470,517

TABLE 5-10: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO C1

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	67 1	$1,130,575 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	61	-	$867,805	$1,205,047	$2,778,847
SE + MNA: 25% Volume Removed	52	-	$976,686	$1,308,583	$2,857,435
SE + MNA: 50% Volume Removed	36	-	$1,131,174	$1,442,596	$2,895,903
SE + MNA: 75% Volume Removed	20	-	$1,186,518	$1,434,976	$2,594,450
SE + MNA: 90% Volume Removed	7	-	$1,117,467	$1,284,323	$2,062,984

TABLE 5-11: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO C2

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	410 1	$1,142,858 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	372	-	$880,904	$1,223,762	$2,823,762
SE + MNA: 25% Volume Removed	314	-	$1,002,260	$1,345,118	$2,945,118
SE + MNA: 50% Volume Removed	217	-	$1,204,520	$1,547,377	$3,147,376
SE + MNA: 75% Volume Removed	120	-	$1,406,261	$1,748,896	$3,347,857
SE + MNA: 90% Volume Removed	41	-	$1,514,618	$1,851,682	$3,424,645

TABLE 5-12: ECONOMIC PROJECTIONS FOR DUAL TECHNOLOGY REMEDIATION ALTERNATIVES - SCENARIO C3

Technology	Years to Remediate	Average Costs	Cost Curves		
MNA SE	1176 1	$1,142,858 $1,536,000	Minimum	Average	Maximum
SE + MNA: 10% Volume Removed	1067	-	$880,904	$1,223,762	$2,823,762
SE + MNA: 25% Volume Removed	900	-	$1,002,260	$1,345,118	$2,945,118
SE + MNA: 50% Volume Removed	620	-	$1,204,520	$1,547,378	$3,147,378
SE + MNA: 75% Volume Removed	341	-	$1,406,780	$1,749,638	$3,349,638
SE + MNA: 90% Volume Removed	118	-	$1,528,130	$1,870,984	$3,470,517

5.6 CHAPTER FIVE FIGURES

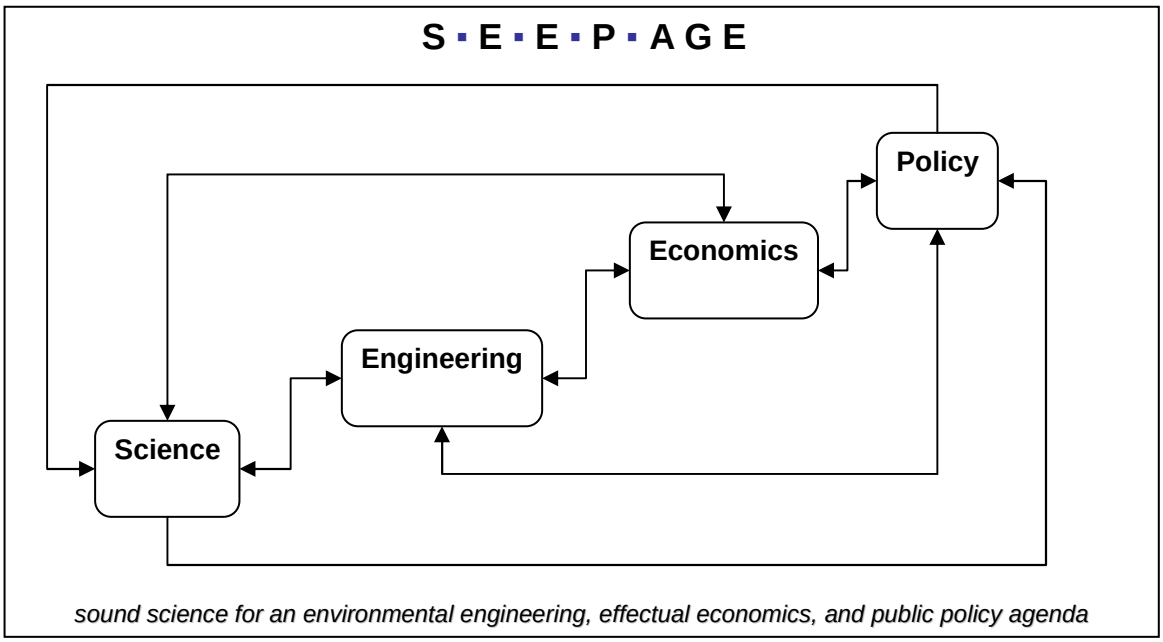
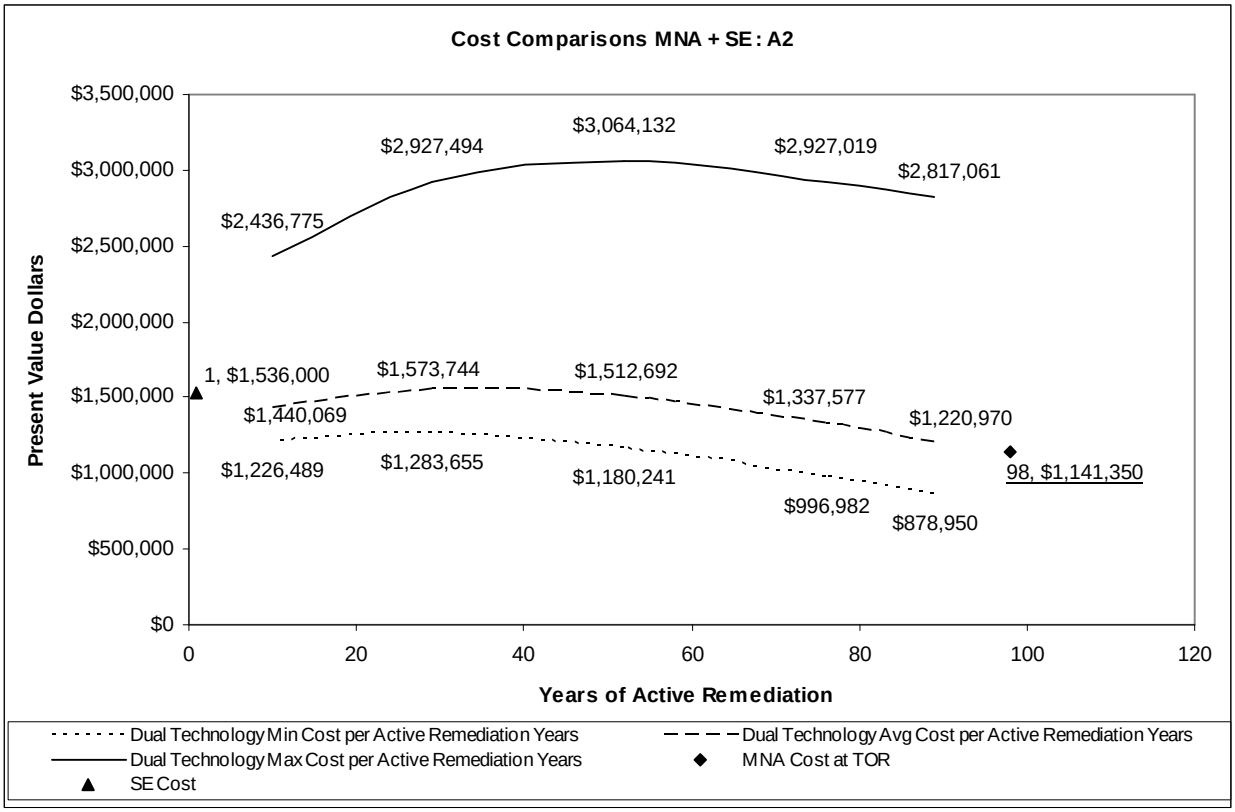


FIGURE 5-1: CONCEPTUAL MODEL OF THE SEEPAGE PARADIGM



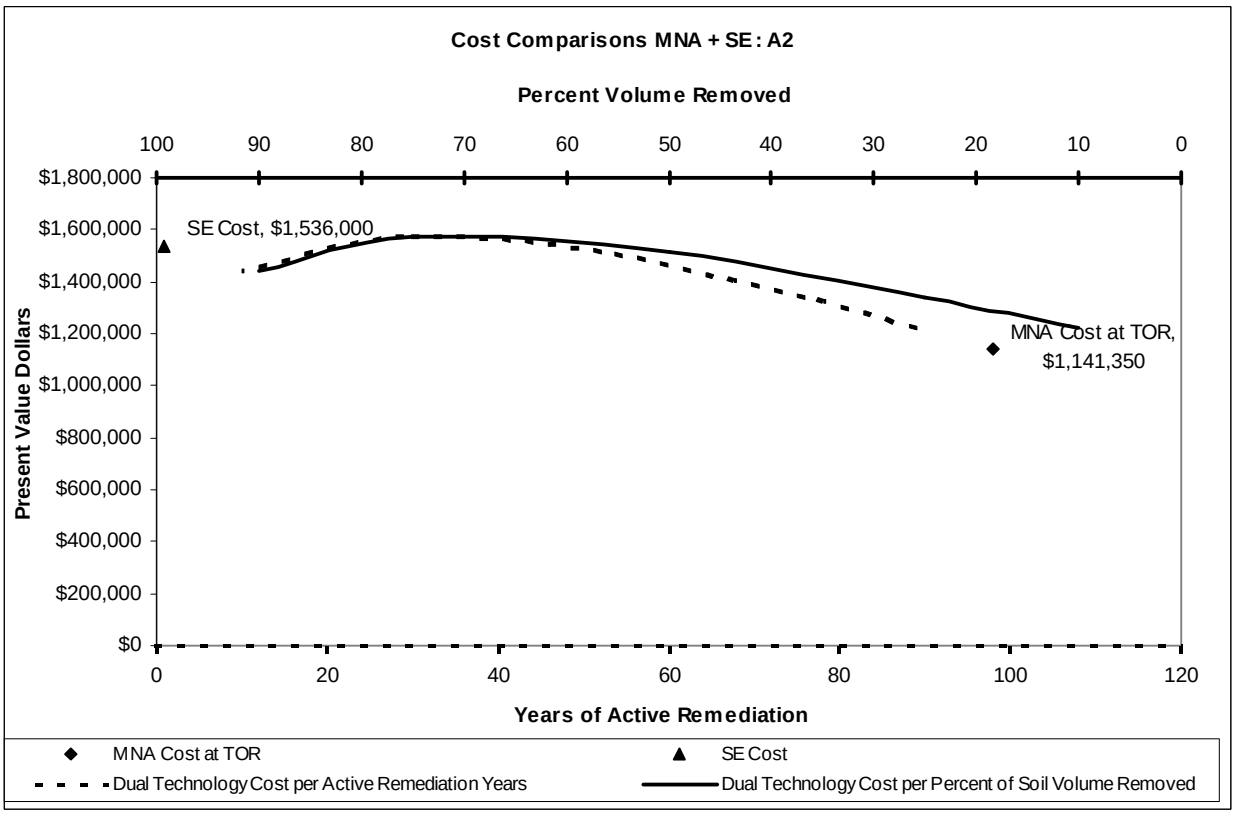
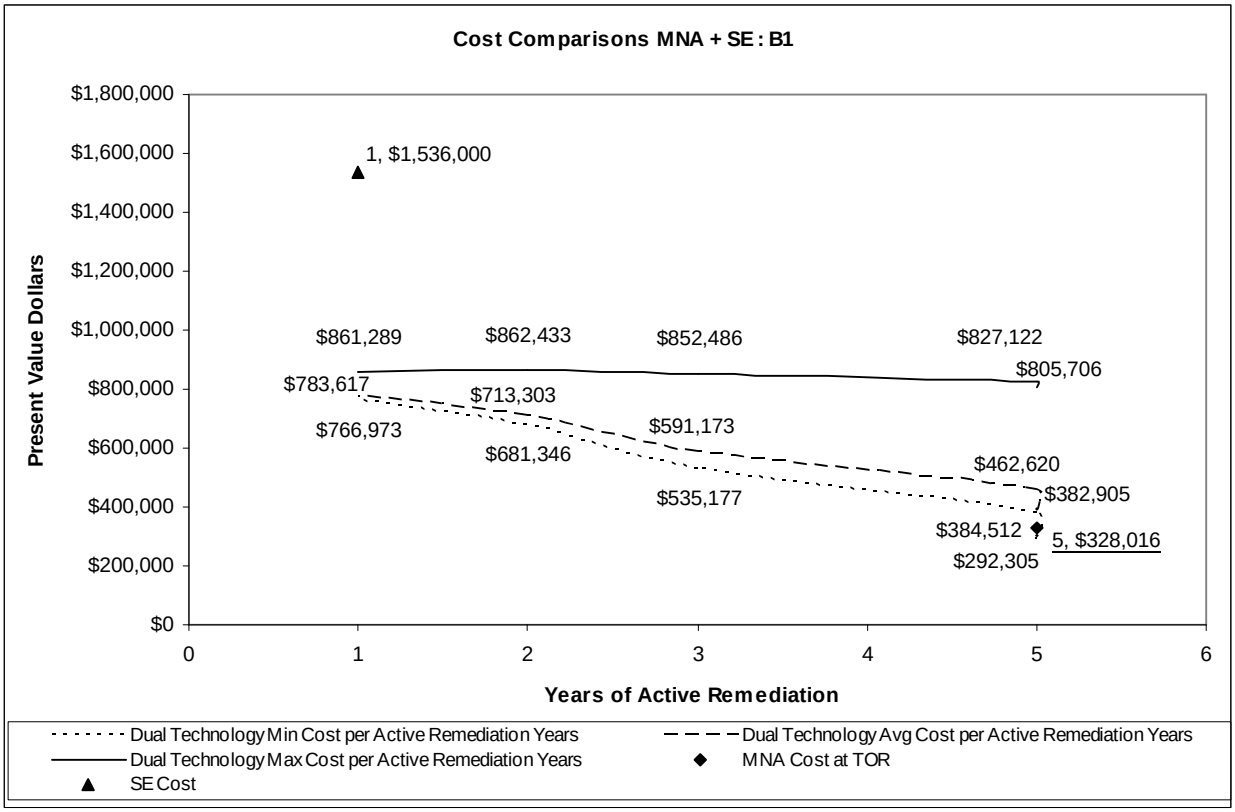


FIGURE 5-2: COMPARISON OF TECHNOLOGY AND CASE STUDY ALTERNATIVES – A2



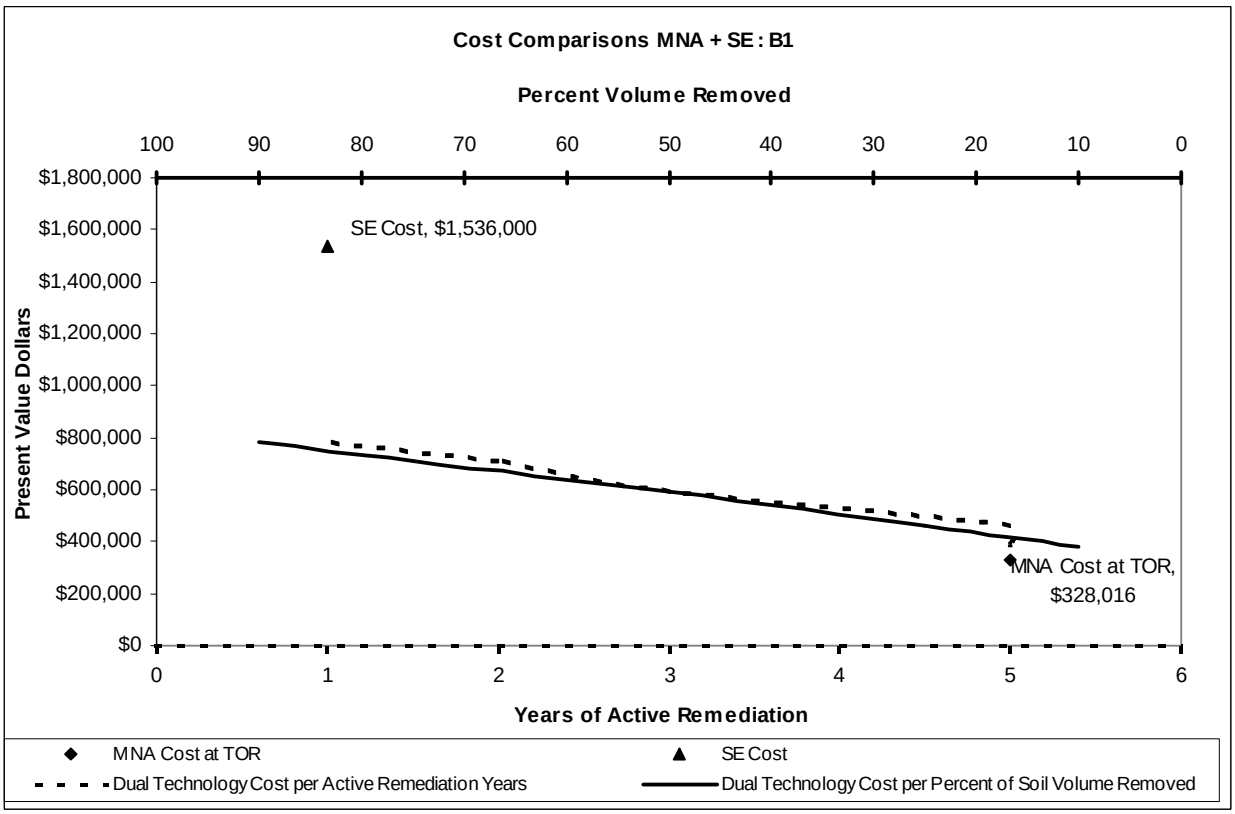


FIGURE 5-3: COMPARISON OF TECHNOLOGY AND CASE STUDY ALTERNATIVES – B1

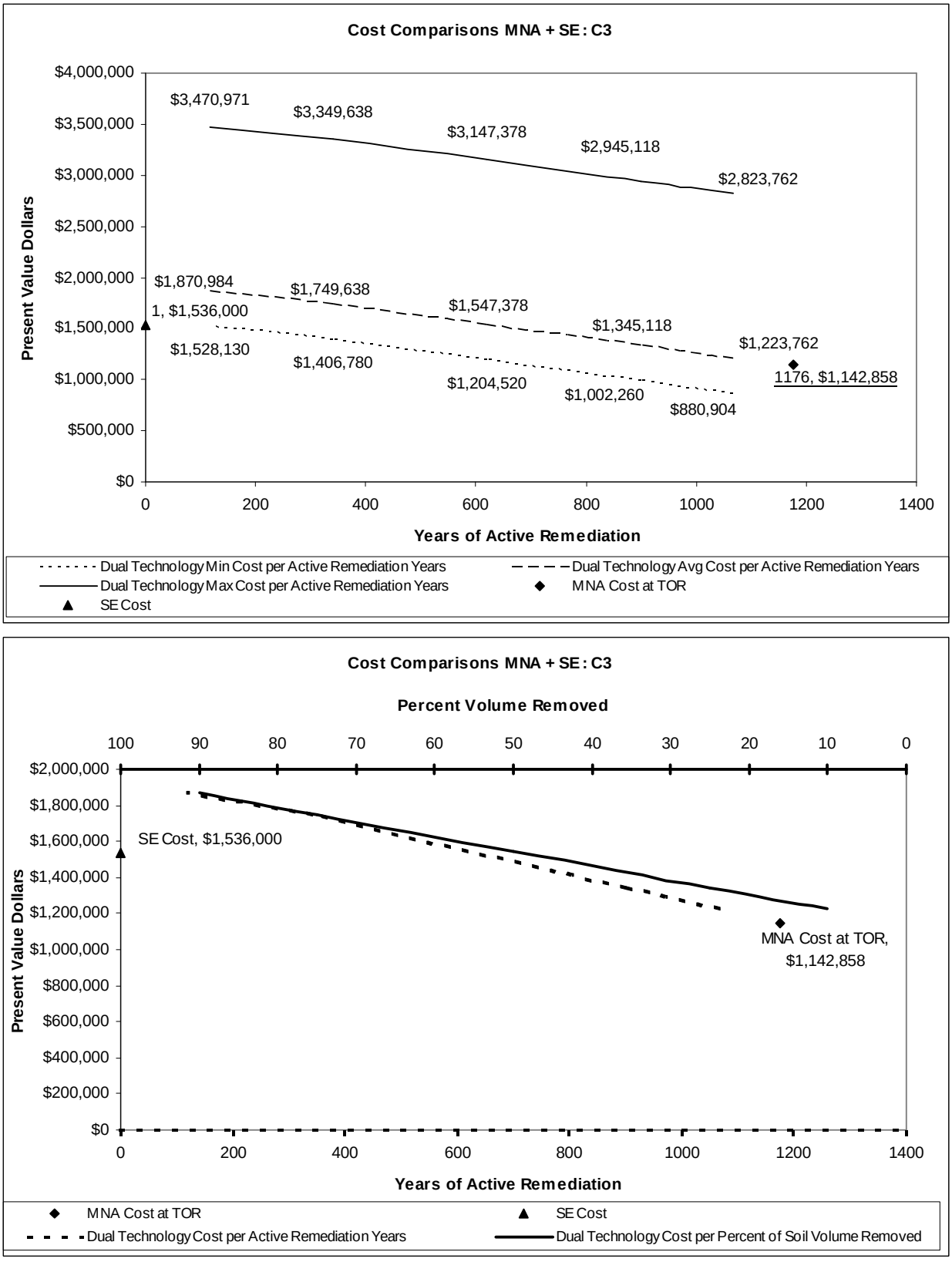


FIGURE 5-4: COMPARISON OF TECHNOLOGY AND CASE STUDY ALTERNATIVES – C3

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