APPENDIX A

SUMMARY OF HISTORICAL GROUNDWATER TRENDS (VOLUME 1)

APPENDIX A

Summary of Historical Groundwater Trends
FUSRAP Maywood Superfund Site
Maywood, New Jersey

1.0 Historical Groundwater Trends

An evaluation of historical groundwater concentration trends over time for selected chemical constituents in overburden and shallow bedrock groundwater wells was performed. The purpose of this evaluation was to determine if concentrations were increasing, decreasing, or remained stable.

Historical groundwater analytical data and RI data were reviewed for FUSRAP Maywood Superfund Site (FMSS) monitoring wells. The selection of groundwater wells and trend analysis was primarily based on the locations of the Maywood Interim Storage Site (MISS) and Areas of Concern (AOCs) 3 and 4, and exceedances of State and Federal MCLs or PRGs. Groundwater monitoring wells within the MISS and AOCs, and upgradient and downgradient of the MISS and AOCs, were selected, if warranted. Groundwater monitoring wells eliminated from the trend analysis were wells with only one sampling event or wells with a high number of non-detected (ND) concentrations. Wells selected that had some ND concentrations had their values reduced by half for trend analysis purposes. In addition, only unfiltered laboratory results were plotted.

The following sections describe the results of the plotted groundwater concentration trends over time for specific chemical constituents of selected overburden and shallow bedrock groundwater wells. The constituent data were reviewed for: lithium, benzene, thallium, arsenic, barium, beryllium, lead, adjusted gross alpha, adjusted gross beta, total radium, total uranium, methylene chloride, tetrachloroethene, toluene, trichloroethene, and vinyl chloride. Refer to Attachment 1 to review the trend plots. Well locations are shown on Groundwater Feasibility Study (GWFS) Figure 4-1.

1.1 Lithium Overburden Groundwater Trends

Eight overburden groundwater wells were selected for lithium trend analysis: MISS02A, B38W01S, MISS06A, MISS05A, B38W19S, MISS03A, B38W15S, and B38W17A.

Trends for northern MISS well MISS02A results show increasing concentrations of lithium through December 2000 (10,100 µg/L), then concentrations decrease in subsequent sampling events to 6,200 µg/L. Well B38W01S had increasing concentrations of lithium until April 1991 (3,550 µg/L), then decreased to a concentration of 1,250 µg/L in the last sampling event (June 2003). In the central portion of the MISS, well MISS06A had a decreasing trend, with concentrations ranging from 12,400 µg/L (July 1991) to less than 100 µg/L (ND for two sampling events). The most recent concentration was 1,450 µg/L. In the southwestern portion of the MISS, wells MISS05A and B38W19S had a decrease in concentration of lithium as well. Well MISS05A ranged from 1,850 µg/L (July 1991) to 560 µg/L (July 2003), and well B38W19S concentrations ranged from 1,730 µg/L (November 2000) to 960 µg/L (July 2003).

Wells downgradient and off the MISS (MISS03A, B38W15S and B38W17A) had trends that decreased in lithium concentration during the last several sampling events after what may have been a longer term increase in concentration. Well MISS03A lithium concentrations ranged from 213 µg/L in October 1990 to less than 100 µg/L (ND) in January 1991. Well B38W15S had increased lithium concentrations from less than 100 µg/L (ND) in January 1991 to 3,240 µg/L (November 2000), then decreasing to 2,240 µg/L in July 2003. Well B38W17A had an increasing and decreasing trend with the highest lithium concentration in July 1991 (551 µg/L), which then decreased to 298 µg/L in June 2001.

1.2 Lithium Shallow Bedrock Groundwater Trends

Eight shallow bedrock groundwater wells were selected for lithium trend plots: MISS02B, B38W18D, MISS01B, MISS05B, MISS07B, MISS03B, B38W15D, and B38W17B. Northern MISS well MISS02B had one of the highest concentrations of lithium (16,700 µg/L), but had a decreasing trend in the last three sampling events with the most recent lithium concentration of 9,590 µg/L. Well B38W18D (south of MISS02B) had a consistent trend of concentrations near 3,000 µg/L with the exception of two lower concentrations of 1,480 µg/L and 307 µg/L. Well MISS01B had a decreasing trend with concentrations ranging from 119 µg/L (October 1990) to ND (less than100 µg/L to less than 38.6 µg/L). Southwest of well MISS02B, well MISS05B increased to 7,400 µg/L in November 2000, then decreased to the most recent concentration of 1,290 µg/L. Well MISS07B had both increases and decreases in the trend data with the lowest concentration of 459 µg/L and the highest concentration of 9,100 µg/L. However, the trend appears to be generally upward.

Just outside the MISS, well MISS03B had an overall decreasing trend with concentrations ranging from 161 µg/L (July 1991) to less than 100 µg/L (ND for two sampling events). Further off the MISS, well B38W15D had an increasing and decreasing trend with the most recent concentration of 2,940 µg/L. Off the MISS, well B38W17B had an overall increasing trend of lithium concentrations ranging from less than 100 µg/L (ND for two sampling events) to 1,970 µg/L (November 2000) with the most recent concentration of 1,810 µg/L.

1.3 Benzene Overburden Groundwater Trends

No overburden groundwater wells were selected for trend analysis due to low detected concentrations of benzene in the MISS overburden groundwater.

1.4 Benzene Shallow Bedrock Groundwater Trends

Seven shallow bedrock wells were selected for benzene trend analysis: MISS02B, MW-3D, MW-26D, BRPZ-9, B38W19D, MISS05B, and MW-24D. Well MISS02B had a decreasing trend with concentrations ranging from 180 µg/L (several sampling events) to very low concentrations. Well MW-3D had a concentration of 5 µg/L (April 2001) and during another sampling event, a concentration of 15 µg/L (August 2002). South of well MISS02B, well MW-26D had concentrations ranging from 520 µg/L (January 2002) to 180 µg/L (August 2002). Further south of well MW-26D, well BRPZ-9 had concentrations of 1,500 µg/L (July 2001) and 1,800 µg/L (August 2002).

In the southwestern portion of the MISS, wells B38W19D and MISS05B had generally low concentrations with an increase to 39 µg/L and 3,500 µg/L, respectively, in June 2001 and November 2000, then decreasing to 0.5 µg/L and less than 5 µg/L (ND), thereafter. In the southern portion of the MISS, well MW-24D had concentrations of 78 μg/L (September 2001) and 29 μg/L (July 2002).

1.5 Thallium Overburden Groundwater Trends

No overburden groundwater wells were selected for trend analysis due to many ND thallium concentrations.

1.6 Thallium Shallow Bedrock Groundwater Trends

Two shallow bedrock groundwater wells were selected in the MISS for thallium trend analysis: B38W02D and B38W18D. Well B38W02D had concentrations ranging from ND (less than 8 μg/L to less than 1 μg/L) to 5.5 μg/L (July 2000). Well B38W18D had concentrations ranging from 7.8 μg/L (July 2000) to ND (less than 50 μg/L to less than 2 μg/L).

1.7 Arsenic Overburden Groundwater Trends

Seven overburden groundwater wells were selected for arsenic trend analysis: MISS02A, MISS06A, B38W19S, MISS03A, B38W14S, B3815S, and B38W17A. All three wells within the MISS, wells MISS02A, MISS06A, and B38W19S, had variable concentrations. Well MISS02A arsenic concentrations ranged from 4,770 μg/L (April 2000) to 2,110 μg/L (July 2002), and well MISS06A arsenic concentrations ranged from 38 μg/L (October 1990) to ND (less than 200 μg/L to less than 2.3 μg/L). Well B38W19S arsenic concentrations ranged from 32 μg/L (November 2000) to ND (less than 200 μg/L to less than 18 μg/L).

Just outside of the MISS, well MISS03A had variable concentrations with the highest concentration being 252 μg/L. Downgradient and off the MISS, wells B38W14S and B38W17A had generally lower concentrations of arsenic. The highest concentrations for wells B38W14S and B38W17A were 20 μg/L and 10.5 μg/L. Well B38W15S had concentrations ranging from 6.4 μg/L to ND (less than 7 μg/L to less than 2 μg/L).

1.8 Arsenic Shallow Bedrock Groundwater Trends

Six shallow bedrock groundwater wells were selected for arsenic trend analysis: B38W18D, MISS07B, MISS05B, B38W19D, MISS03B, and B38W15D. Within the eastern portion of the MISS, well B38W18D had concentrations of arsenic ranging from 23 μg/L (April 2000) to ND (less than 10 μg/L to less than 2 μg/L).

In the western portion of the MISS, well MISS07B showed wide ranges of concentrations; well MISS05B had a generally increasing trend; and well B38W19D had an overall decreasing trend during the later sampling period. MISS07B had the highest concentration of arsenic of the three wells with concentrations ranging from 155 μg/L (July 1991) to 4.6 μg/L (January 1991).

Just outside the MISS, well MISS03B had concentrations of arsenic with concentrations ranging from 10.3 μg/L (April 1991) to ND (less than 6.1 μg/L to less than 2 μg/L).

Downgradient and off the MISS, well B38W15D had arsenic concentrations ranging from 19.1 μg/L (November 2000) to less than 2 μg/L (ND for two sampling events).

1.9 Barium Overburden Groundwater Trends

There were no overburden wells selected for trend analysis due to low concentrations of barium within the MISS.

1.10 Barium Shallow Bedrock Groundwater Trends

Four MISS shallow bedrock groundwater wells were selected for trend analysis: B38W25D, MISS05B, B38W24D, and B38W15D. All wells, with the exception of B38W25D, had a decreasing trend with low concentrations of barium with the exception of the last sampling event. Well B38W25D had an increase in barium concentrations with the highest concentration of 312 µg/L. The last sampling event had a lower result of 237 µg/L.

1.11 Beryllium Overburden Groundwater Trends

There were no overburden wells selected for trend analysis due to low concentrations of beryllium.

1.12 Beryllium Shallow Bedrock Groundwater Trends

There were no shallow bedrock wells selected for trend analysis due to low concentrations of beryllium.

1.13 Lead Overburden Groundwater Trends

Four overburden groundwater wells were selected for lead trend analysis: MISS02A, MISS06A, B38W14S, and B38W17A. Wells MISS02A and MISS06A had overall decreasing trends. Well MISS02A concentrations ranged from 30 µg/L (January 1991) to ND (less than 3.6 µg/L to less than 50 µg/L). The highest concentration for well MISS06A was 79 µg/L (October 1990) and less than 3.6 µg/L (ND) in the last sampling event.

Downgradient and off the MISS, wells B38W14S and B38W17A also had decreasing trends with lead concentrations of 62.4 µg/L for B38W14S and 168 µg/L for well B38W17A in 1991 to ND (less than 10 µg/L to less than 1.1 µg/L) for both wells thereafter.

1.14 Lead Shallow Bedrock Groundwater Trends

There were no shallow bedrock wells selected for trend analysis due to low lead concentrations within the MISS.

1.15 Adjusted Gross Alpha Overburden Groundwater Trends

There were no overburden groundwater wells selected for trend analysis due to many NDs and laboratory-reported measurement errors.

1.16 Adjusted Gross Alpha Shallow Bedrock Groundwater Trends

There were no shallow bedrock wells selected for trend analysis due to many NDs and laboratory-reported measurement errors.

1.17 Adjusted Gross Beta Overburden Groundwater Trends

There were no overburden wells selected for trend analysis due to many NDs and laboratory-reported measurement errors.

1.18 Adjusted Gross Beta Shallow Bedrock Groundwater Trends

There were no shallow bedrock wells selected for trend analysis due to many NDs and laboratory-reported measurement errors.

1.19 Total Radium Overburden Groundwater Trends

Five overburden groundwater wells were selected for trend analysis of total radium: B38W19S, MISS06A, MISS03A, B38W14S, and B38W15S. In the south-central portion of the MISS, well B38W19S had an increasing trend with total radium concentrations of 0.59 pCi/L (May 1994) to 5.18 pCi/L (July 2003). Well MISS06A had concentrations ranging from 0.7 pCi/L (July 2002) to 6.30 pCi/L (July 2003).

Just outside the MISS, well MISS03A had a decreasing trend with concentrations ranging from 7.5 pCi/L (July 1991) to 1.0 pCi/L (July 2001). Downgradient and off the MISS, wells B38W14S and B38W15S had overall decreasing trends that were more recently near 1.0 pCi/L.

1.20 Total Radium Shallow Bedrock Groundwater Trends

Seven shallow bedrock groundwater wells were selected for trend analysis of total radium: MISS02B, B38W18D, MISS07B, MISS05B, MISS03B, B38W04B, and B38W15D. Within the MISS, wells MISS02B, B38W18D, MISS07B, and MISS05B all had decreasing trends with current concentrations near 1.0 pCi/L.

Just outside the MISS, well MISS03B had a decreasing trend with low concentrations of total radium, and well B38W04B was variable with the most current result being less than 2 pCi/L.

Downgradient of the MISS, well B38W15D had a decreasing trend with the last six sample results being less than 2 pCi/L.

1.21 Total Uranium Overburden Groundwater Trends

Six overburden groundwater wells were selected for trend analysis of total uranium: B38W19S, B38W25S, MISS05A, MISS03A, B38W15S, and B38W17A. Within the MISS, well B38W19S had increasing and decreasing concentrations with generally lower concentrations of total uranium. Well B38W25S had a decreasing trend with one total uranium concentration near 8 µg/L (July 2001), while all other concentrations were below 2 µg/L. Well MISS05A had increasing and decreasing concentrations with values ranging from 75 µg/L (June 2001) to 544 µg/L (June 1998).

Just outside of the MISS, well MISS03A had increasing and decreasing total uranium concentrations ranging from 15 µg/L (July 1991) to 0.4 µg/L (November 2000). Downgradient of the MISS, wells B38W15S and B38W17A both had overall decreasing trends with the most recent concentrations less than 2.0 µg/L.

1.22 Total Uranium Shallow Bedrock Groundwater Trends

There were no shallow bedrock wells selected for trend analysis due to low concentrations of total uranium.

1.23 Methylene Chloride Overburden Groundwater Trends

There were no overburden groundwater wells selected for trend analysis due to the low number of detected concentrations of methylene chloride.

1.24 Methylene Chloride Shallow Bedrock Groundwater Trends

There were no shallow bedrock groundwater wells selected for trend analysis due to the low number of detected concentrations of methylene chloride.

1.25 Tetrachloroethene Overburden Groundwater Trends

There were no overburden groundwater wells selected for trend analysis due to the low number of detected concentrations of tetrachloroethene.

1.26 Tetrachloroethene Shallow Bedrock Groundwater Trends

Four shallow bedrock wells were selected for groundwater trend analysis of tetrachloroethene: MISS02B, MISS05B, MISS01B, and MISS07B. Within the MISS, wells MISS02B and MISS05B had decreasing trends with higher initial tetrachloroethene concentrations (near 30 μg/L) that decreased to ND (less than 12.5 μg/L to less than 0.1 μg/L). MISS wells MISS01B and MISS07B had increasing and decreasing concentrations. Well MISS01B concentrations ranged from ND (less than 5 μg/L) to 140 μg/L (May 1994). Well MISS07B concentrations ranged from 110 μg/L (January 1985) to 2 μg/L (July 2003).

1.27 Toluene Overburden Groundwater Trends

There were no selected overburden wells for trend analysis of toluene due to low concentrations.

1.28 Toluene Shallow Bedrock Groundwater Trends

There were no selected shallow bedrock wells for trend analysis of toluene due to low concentrations.

1.29 Trichloroethene Overburden Groundwater Trends

There were no selected overburden wells for trend analysis of trichloroethene due to low concentrations of trichloroethene within the MISS.

1.30 Trichloroethene Shallow Bedrock Groundwater Trends

Two shallow bedrock groundwater wells were selected for trichloroethene trend analysis: MISS01B and MISS07B. Within the MISS, well MISS01B had variable concentrations with trichloroethene concentrations ranging from less than 5 μg/L (ND for several sampling events) to 9 μg/L (May 1994 and May 1996). Well MISS07B had a decreasing trend with concentrations ranging from 16 μg/L (January 1986) to less than 5 μg/L (ND for several sampling events).

1.31 Vinyl Chloride Overburden Groundwater Trends

There were no selected overburden wells for trend analysis of vinyl chloride due to low concentrations of vinyl chloride within the MISS.

1.32 Vinyl Chloride Shallow Bedrock Groundwater Trends

There were no selected shallow bedrock wells for trend analysis of vinyl chloride due to low concentrations of vinyl chloride within the MISS.

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ATTACHMENT 1

GROUNDWATER TREND PLOTS

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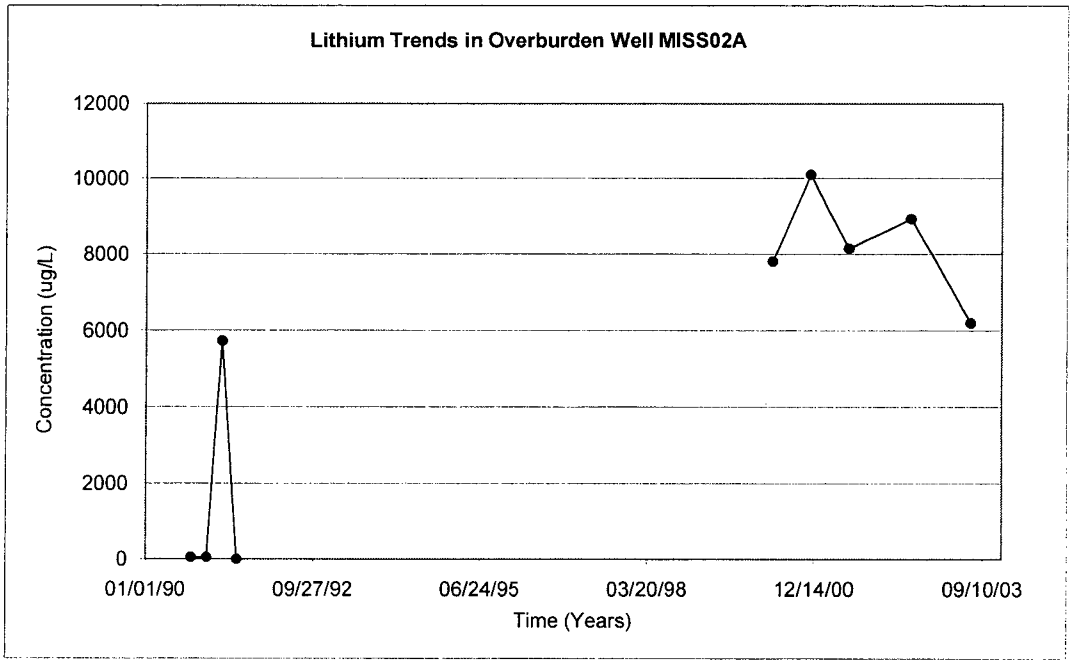
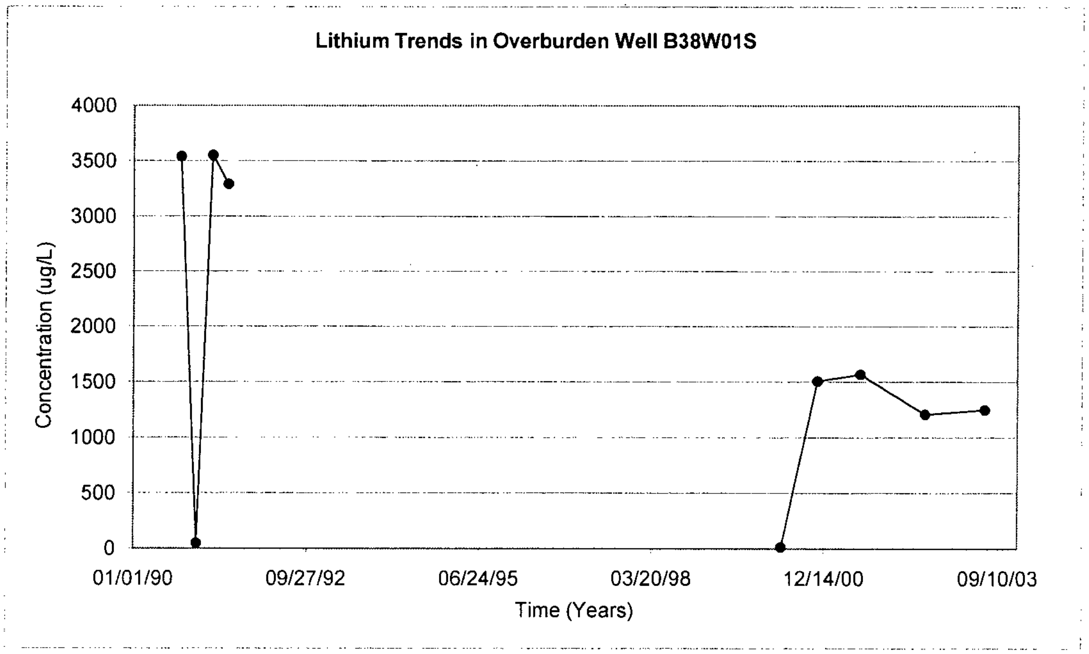
MAYWOOD, NEW JERSEY





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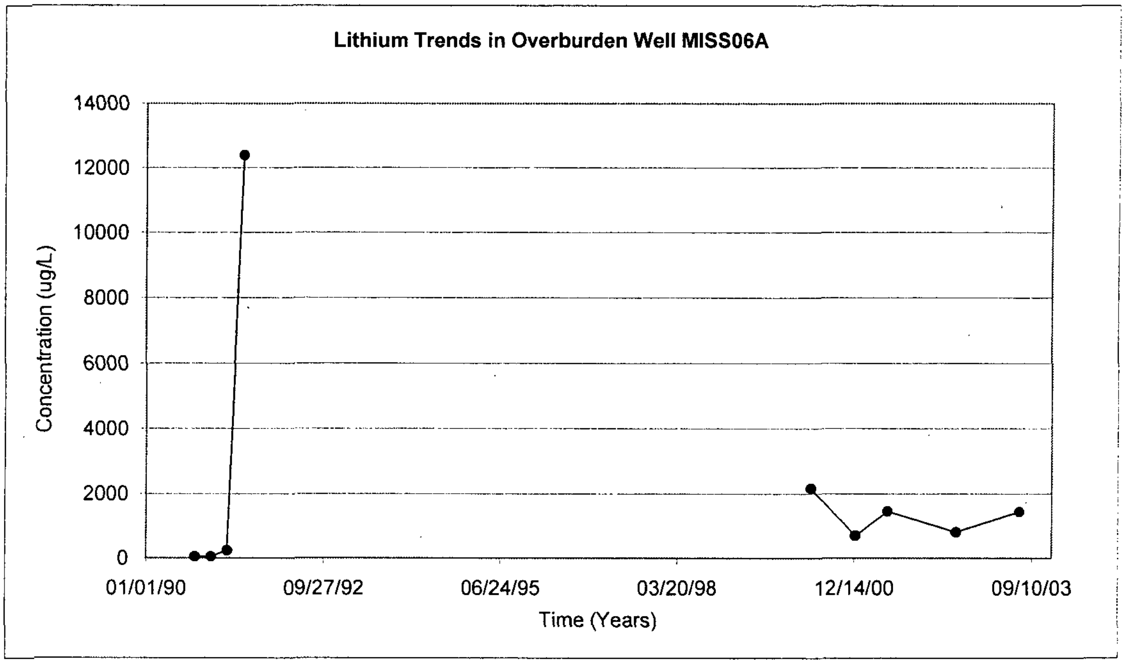

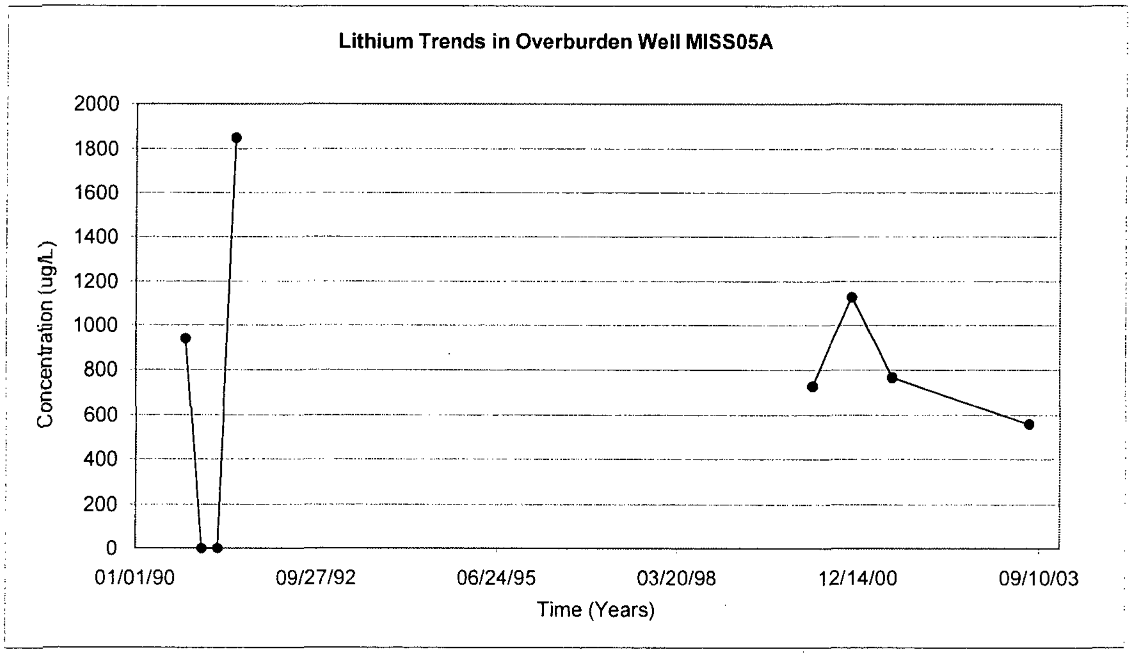

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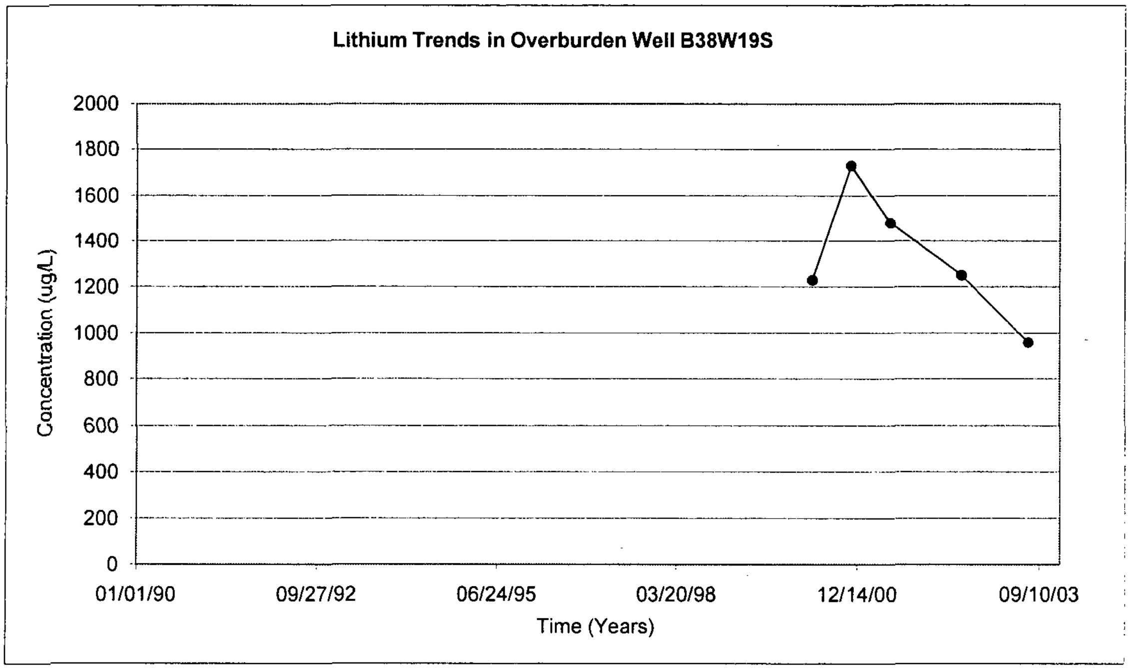

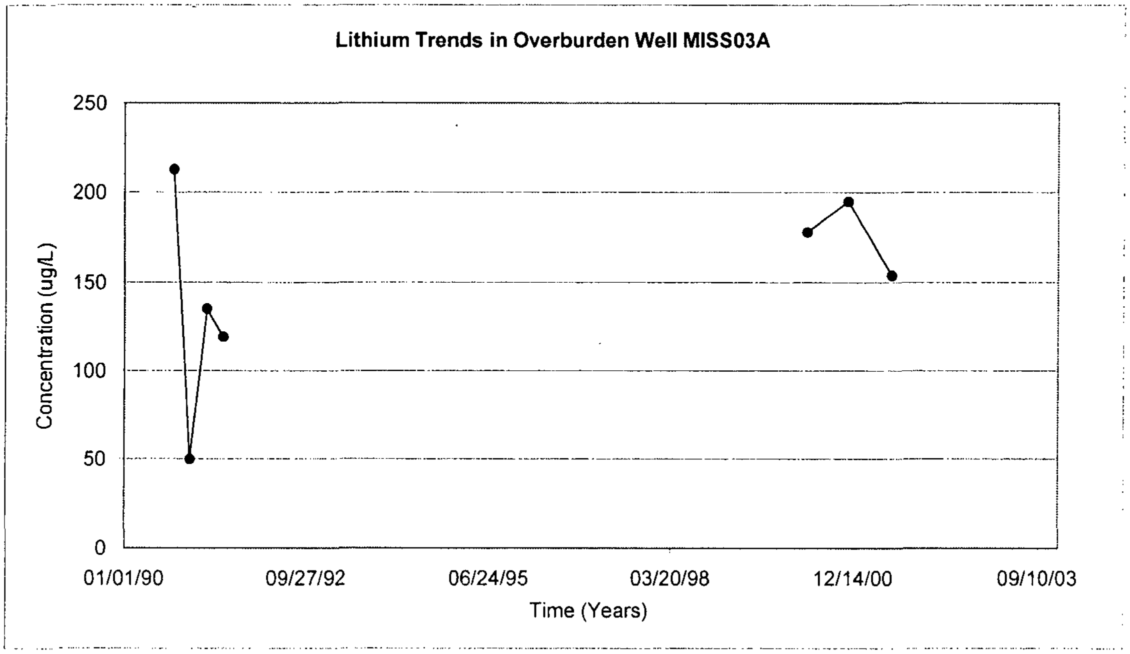

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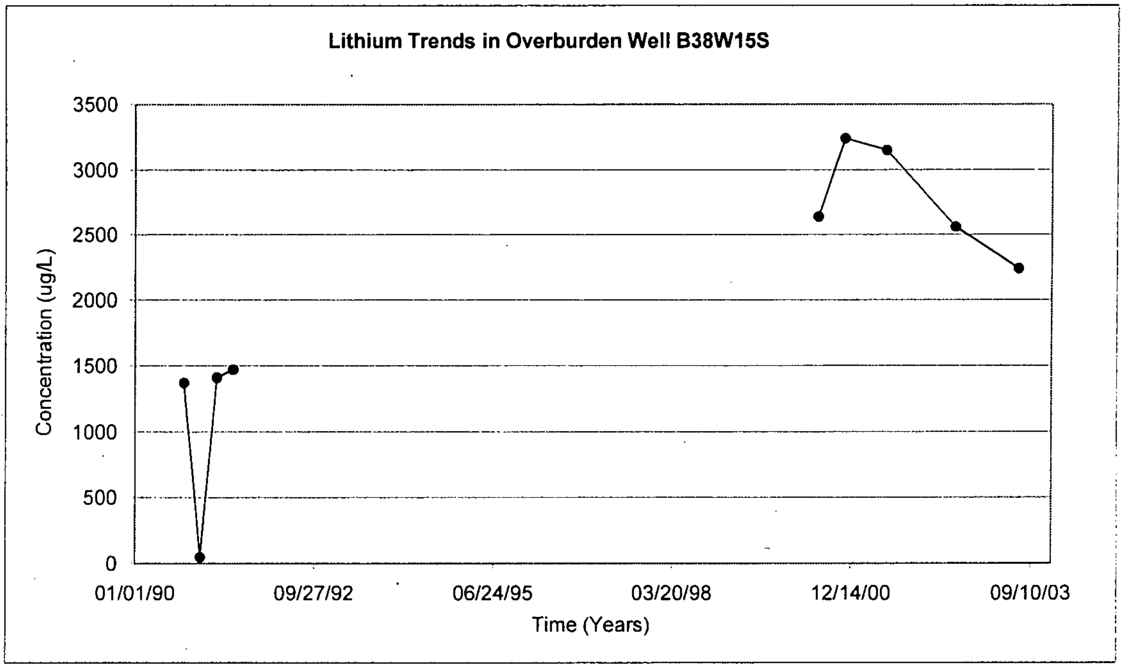

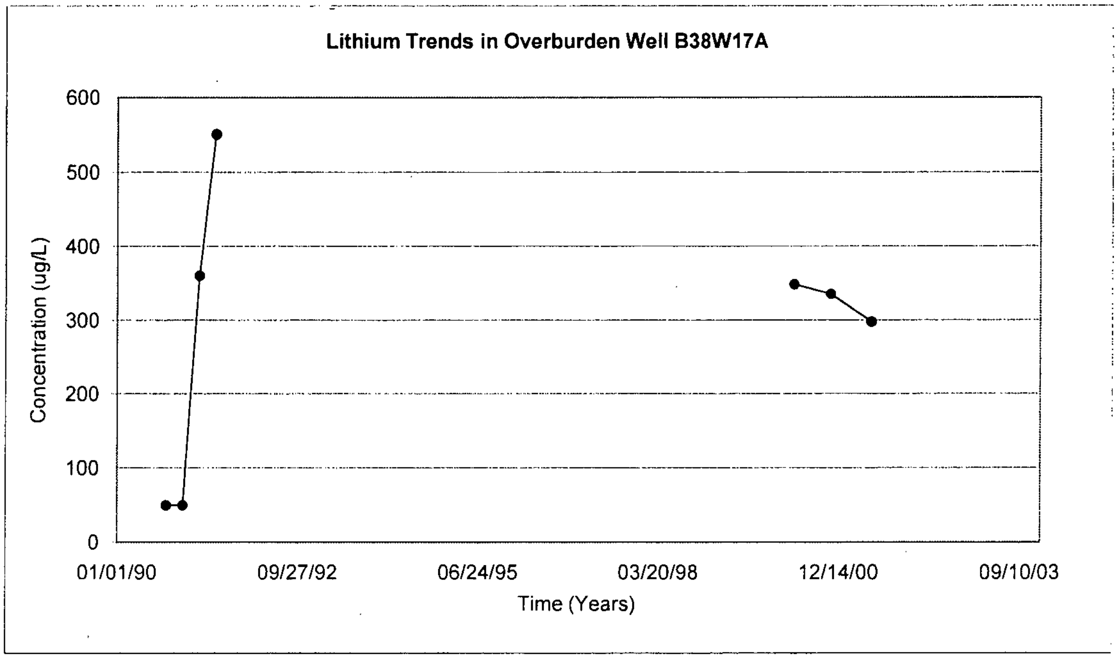

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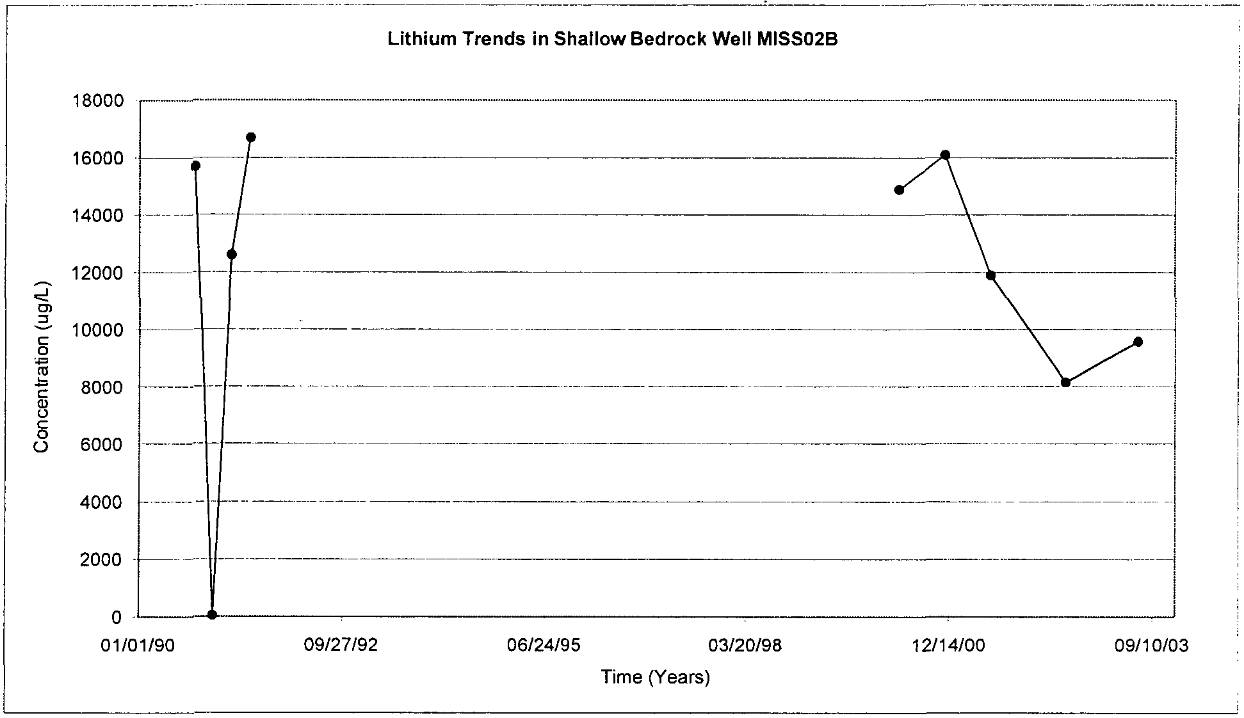

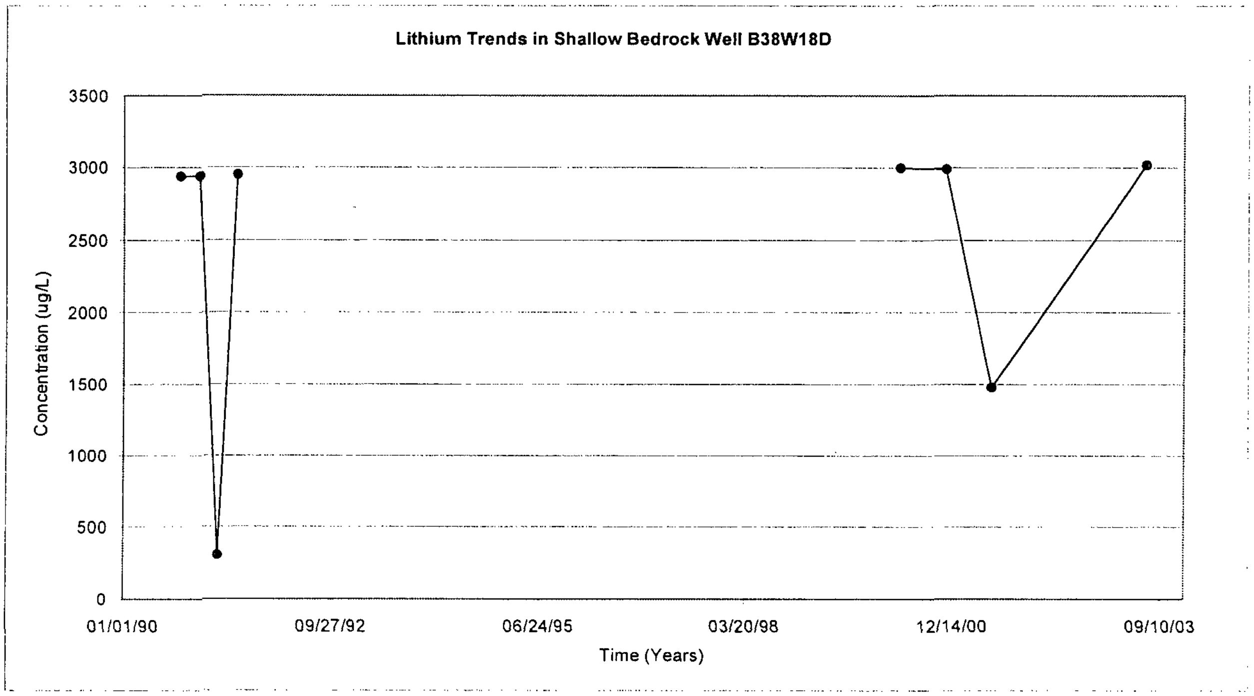

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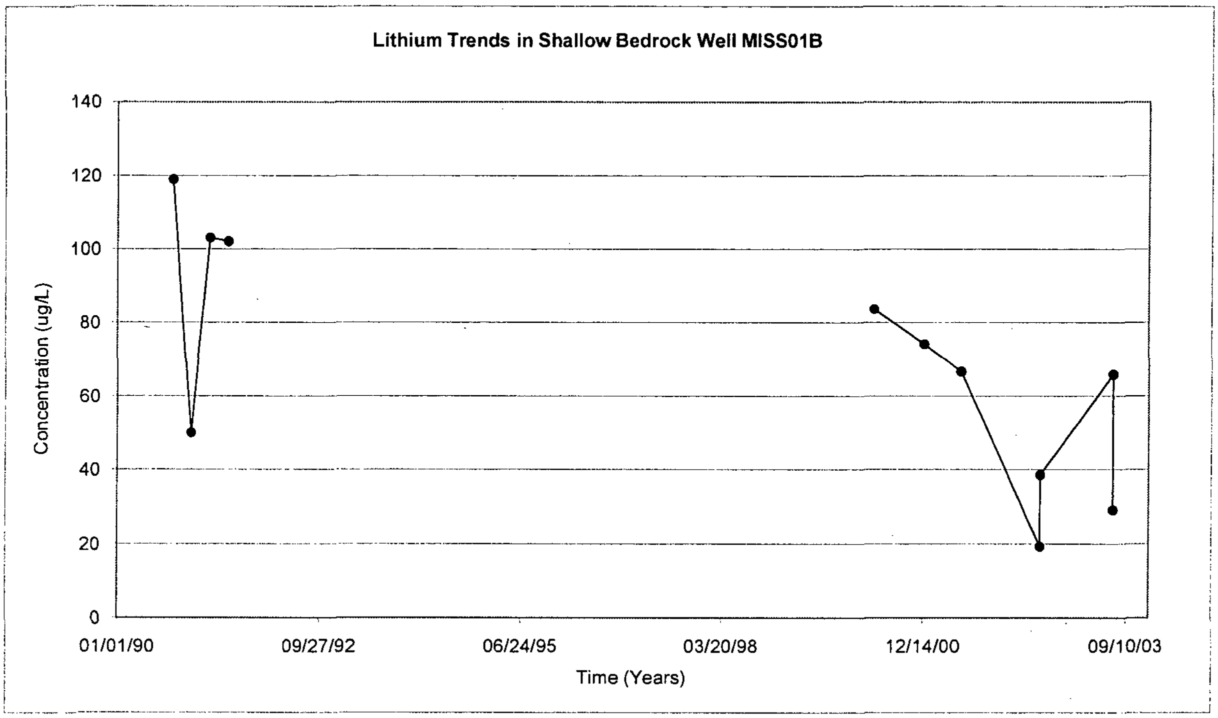

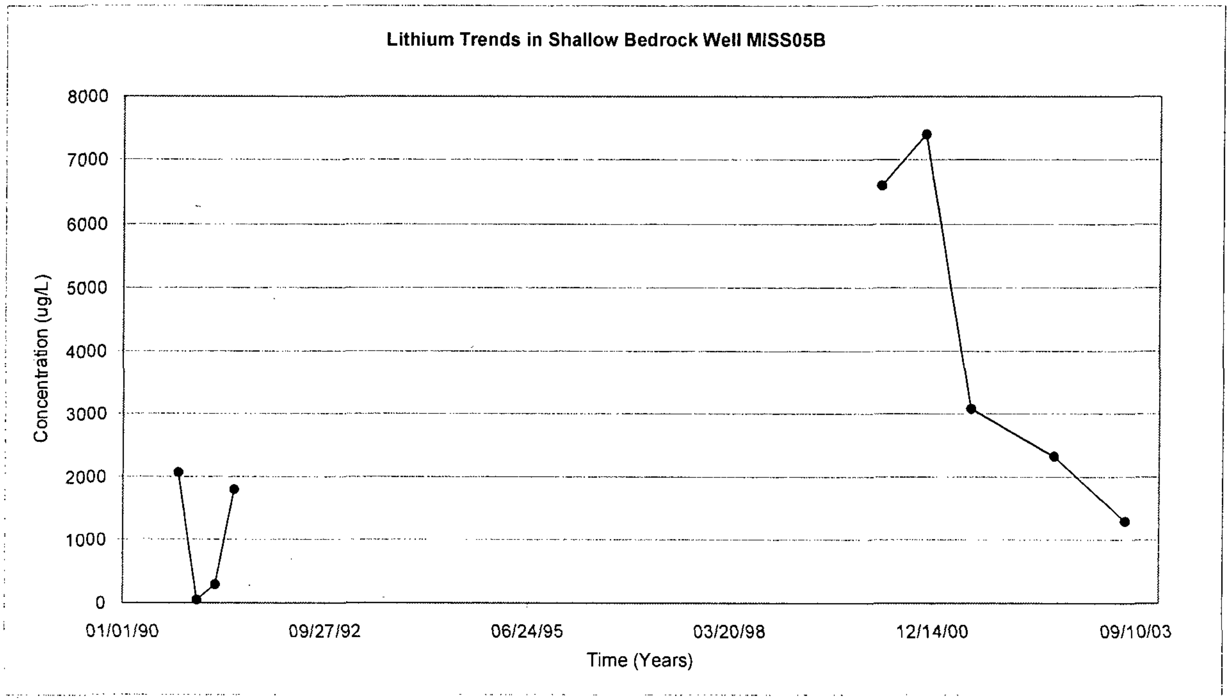

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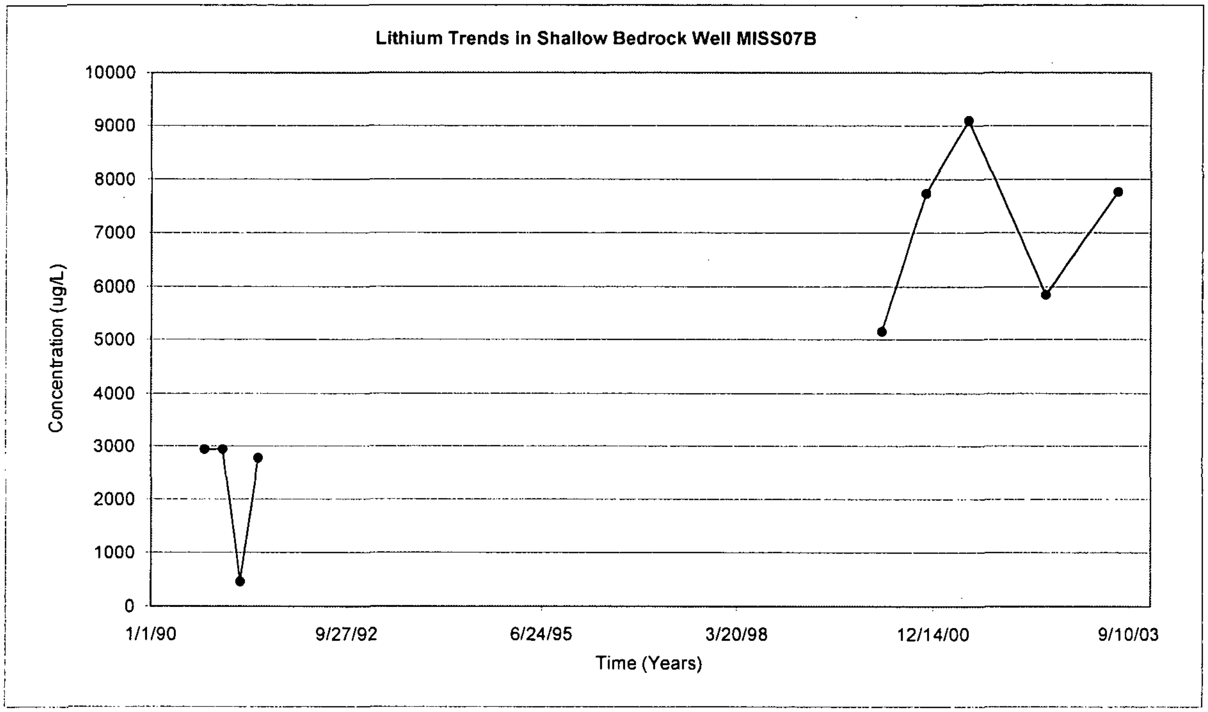

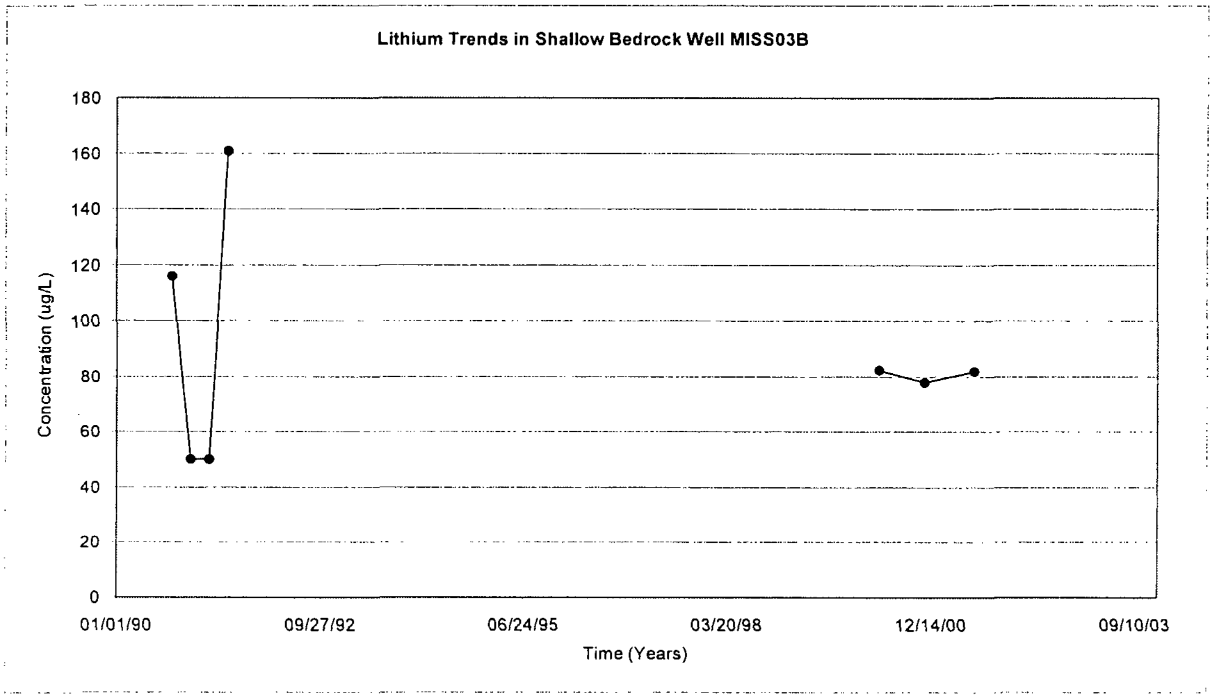

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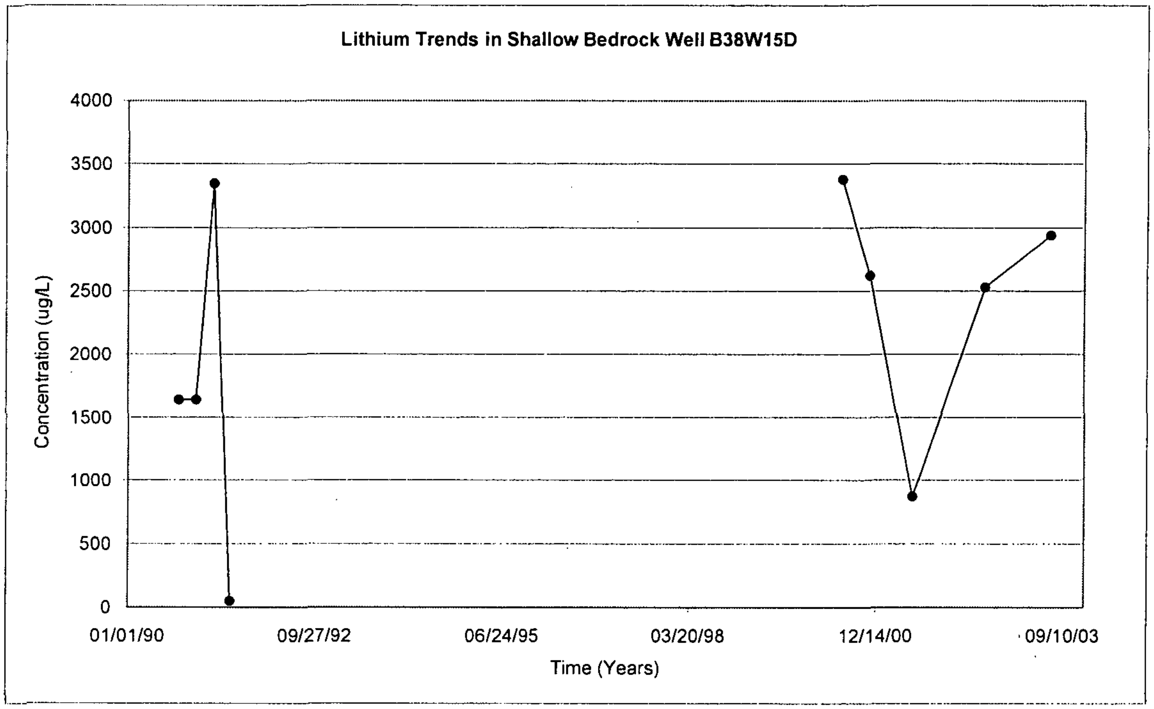

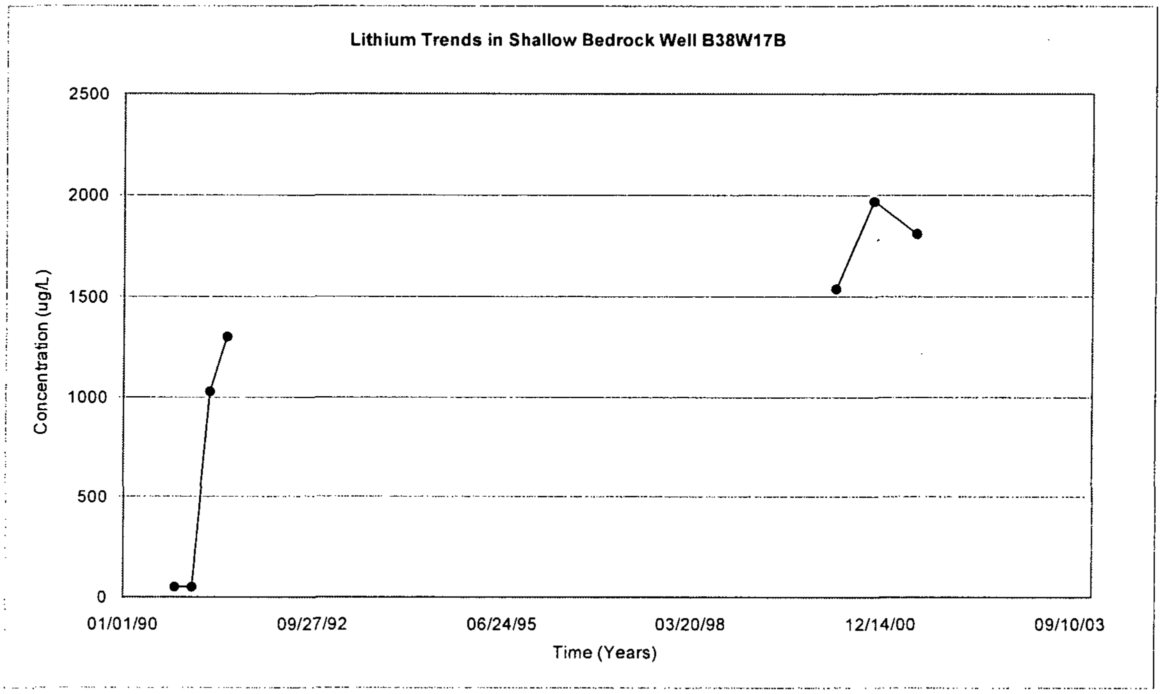

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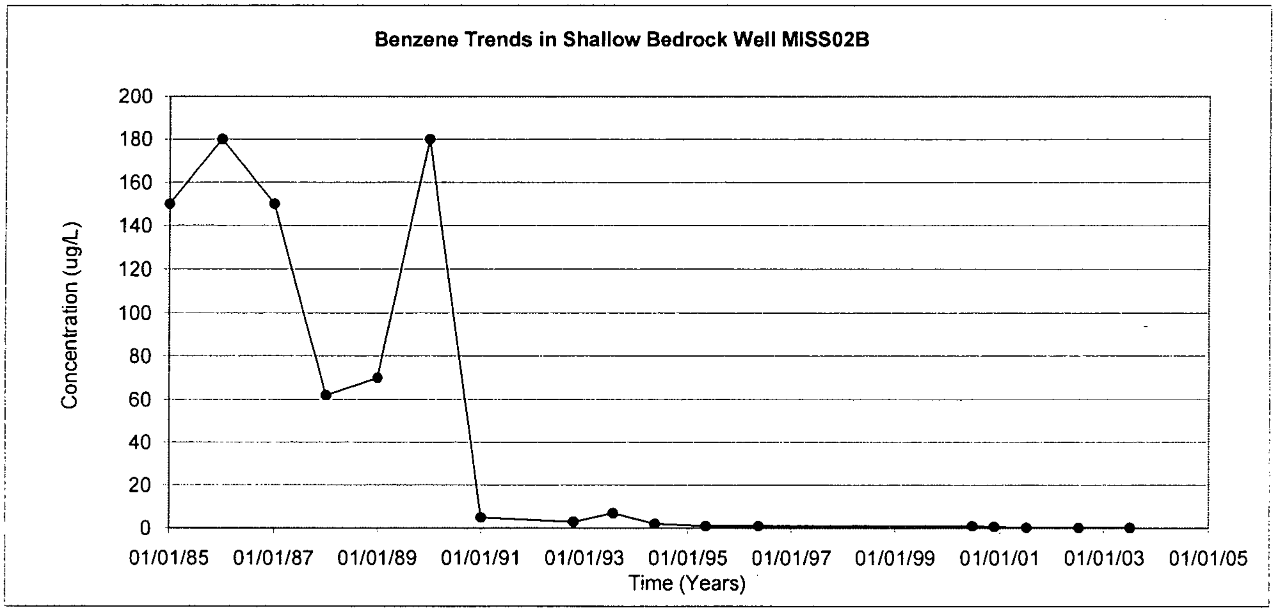

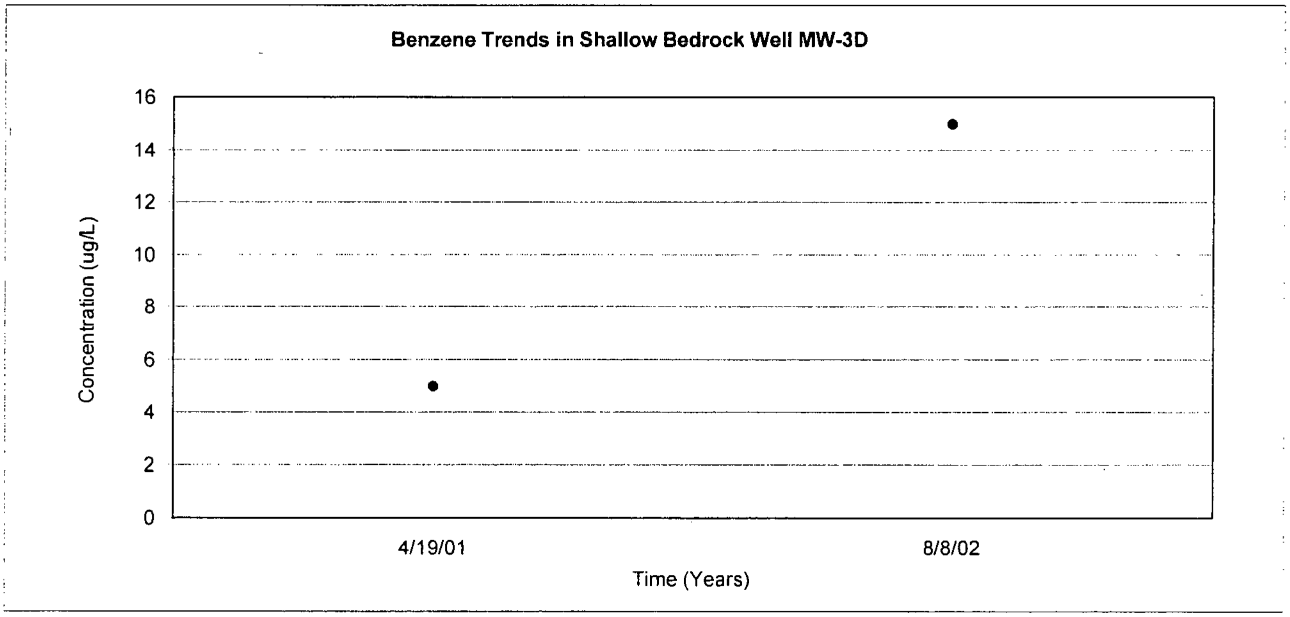

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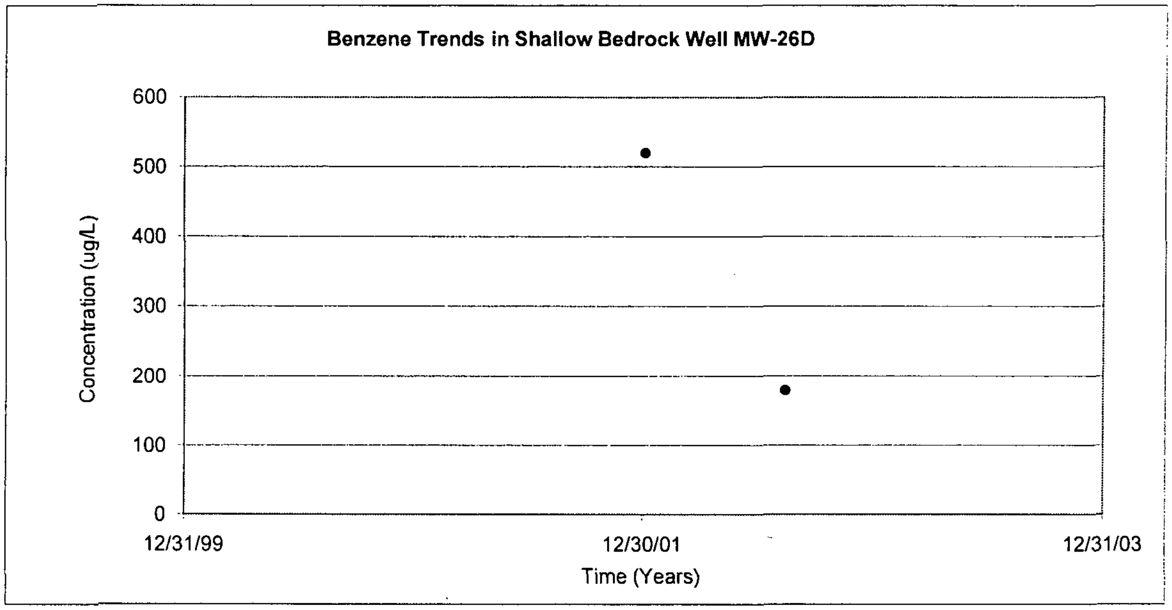

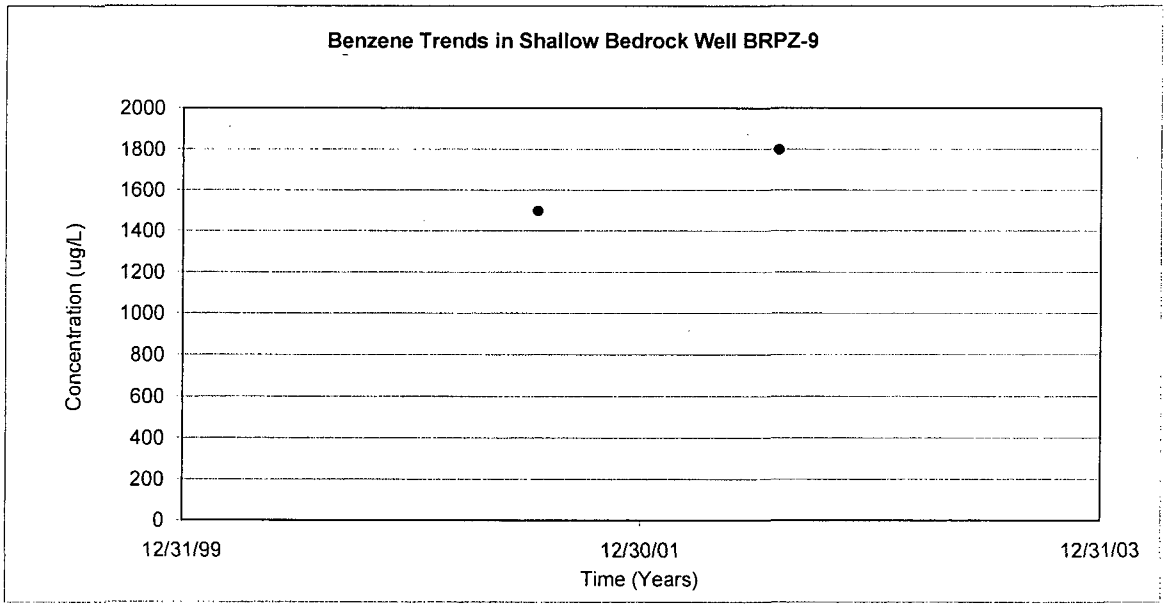

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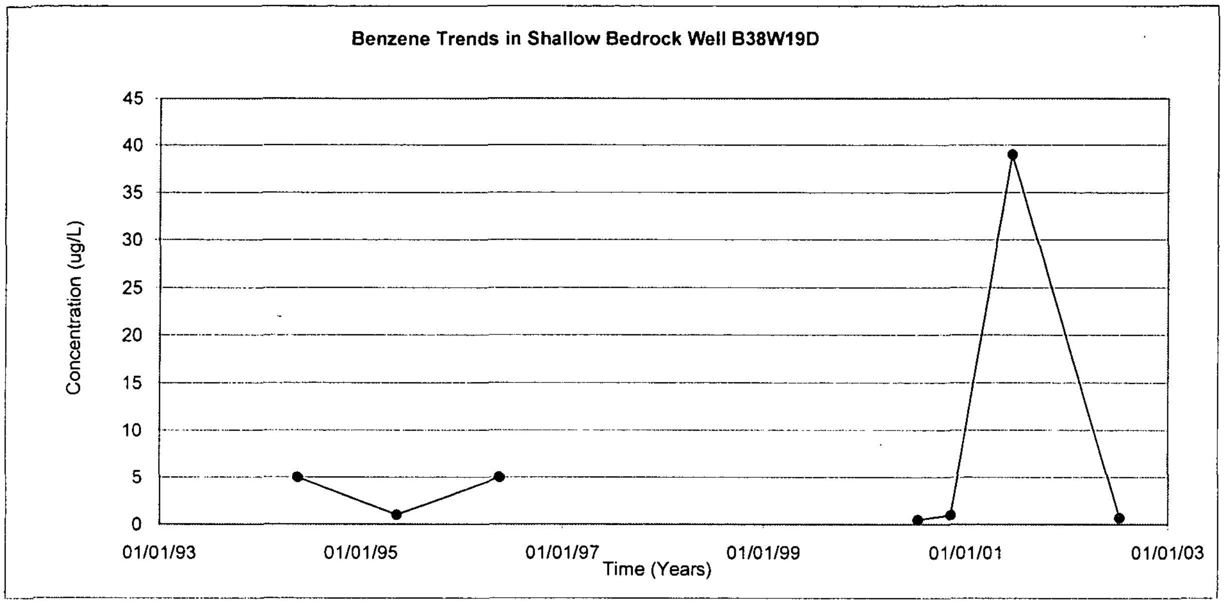

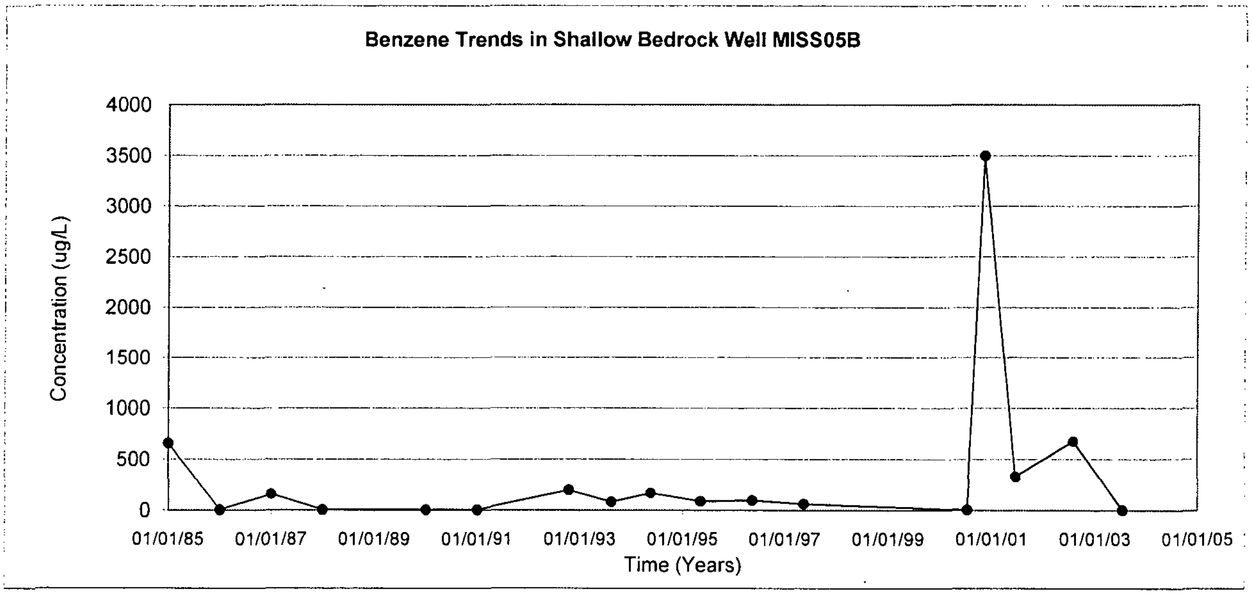

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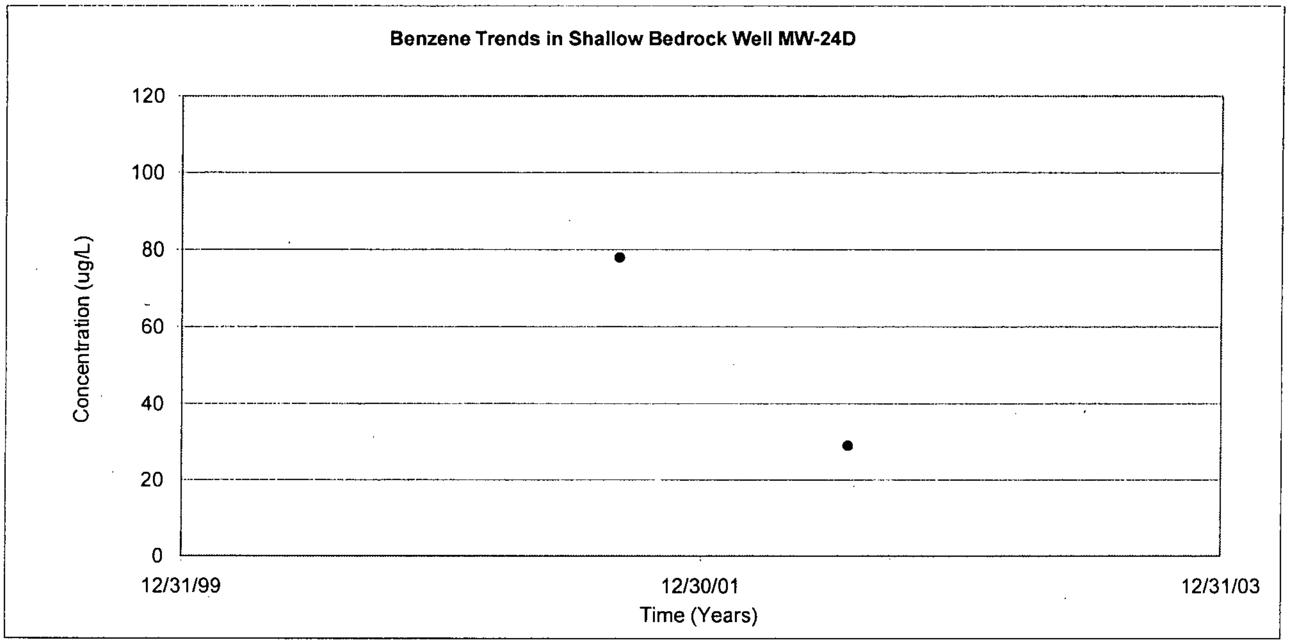

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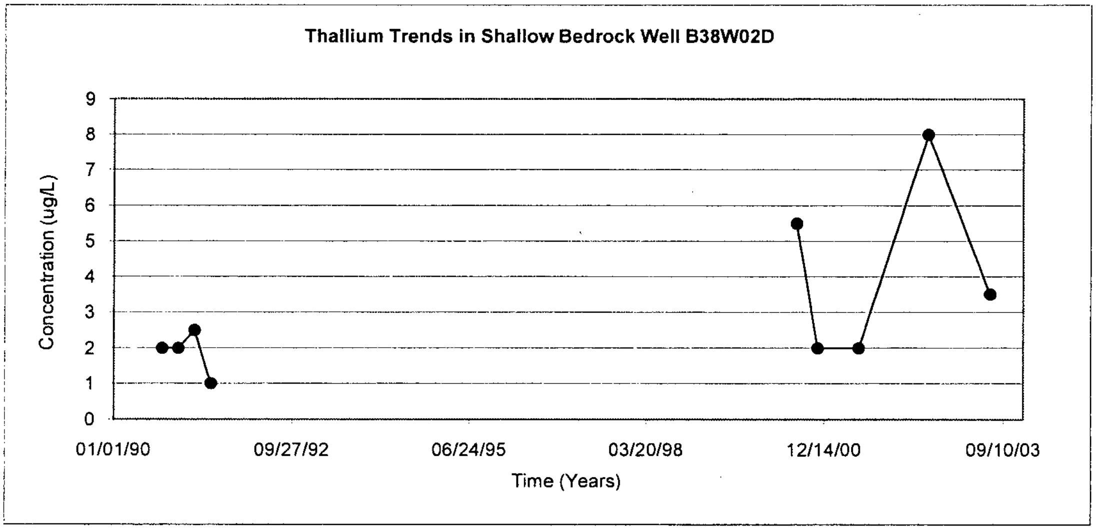

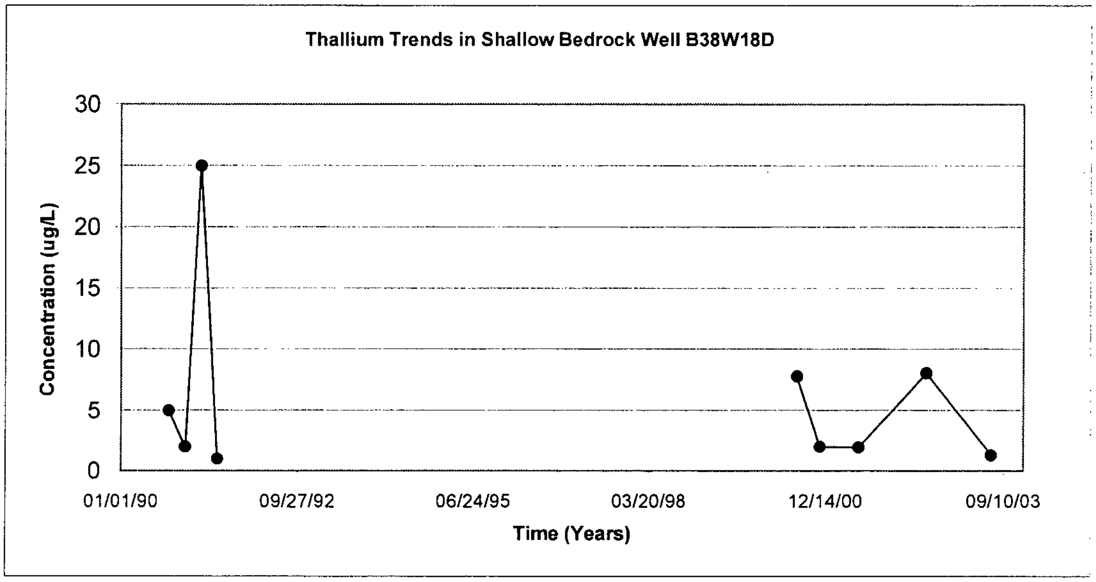

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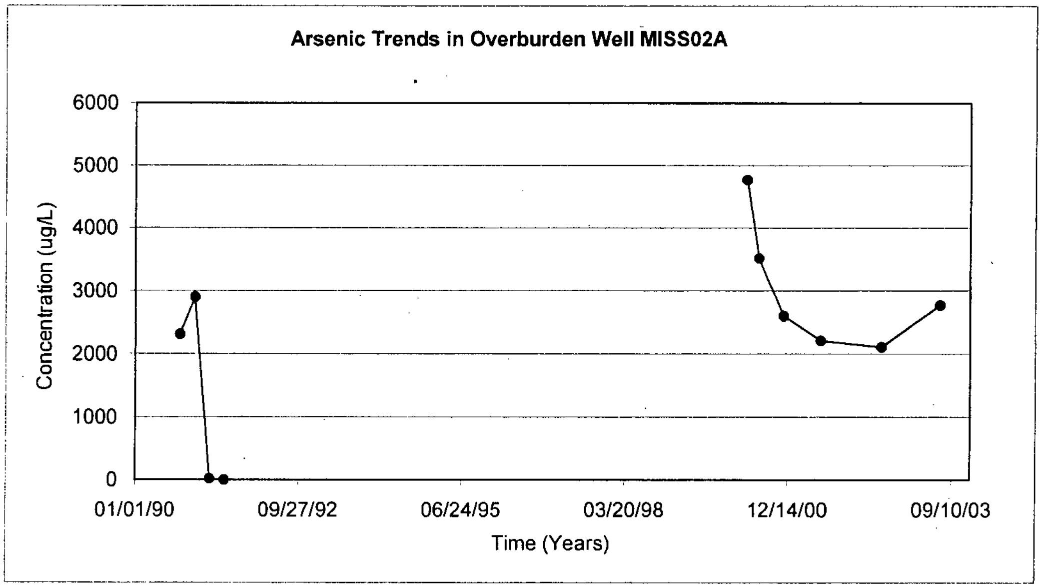

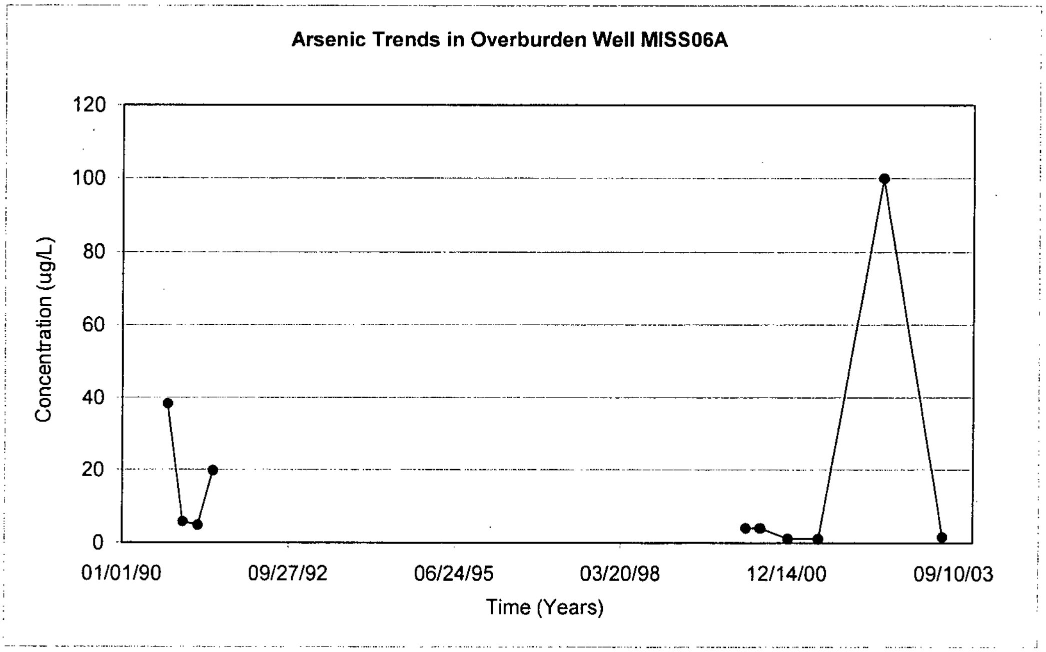

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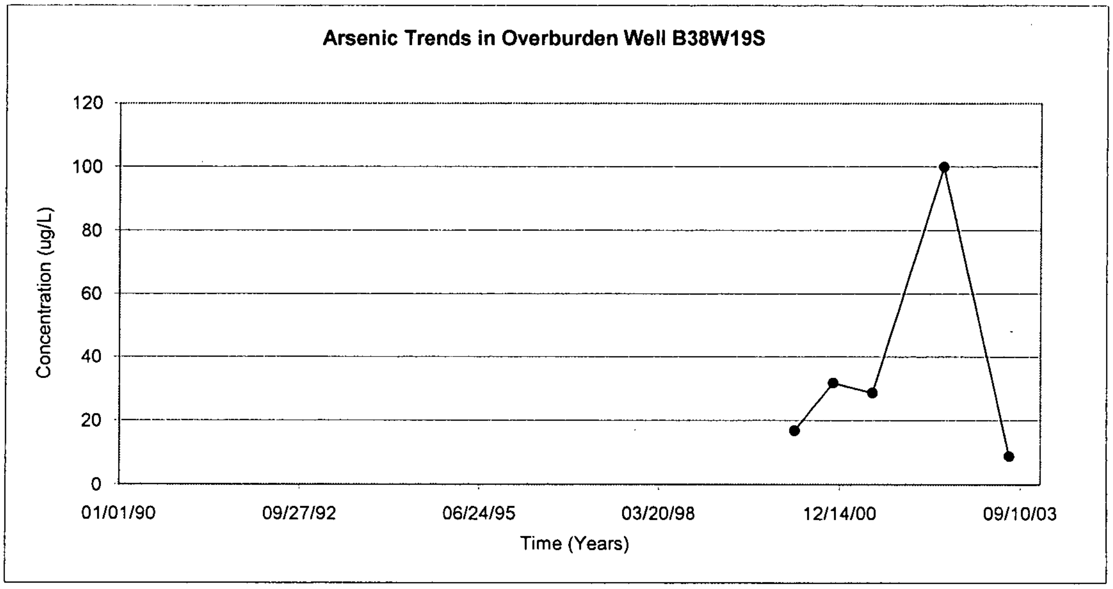

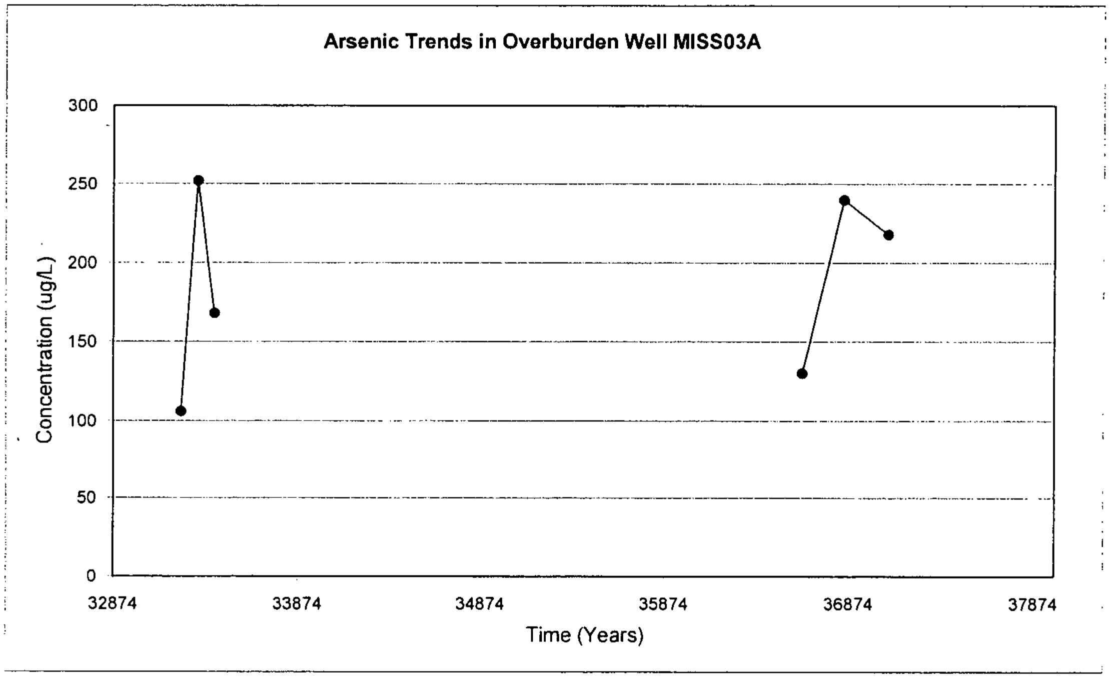

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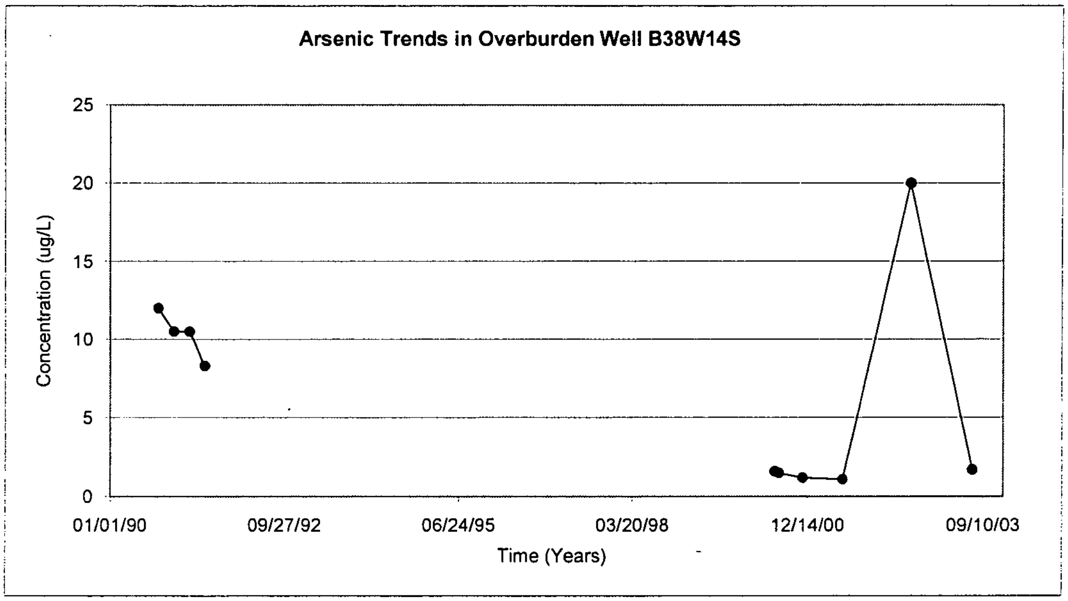

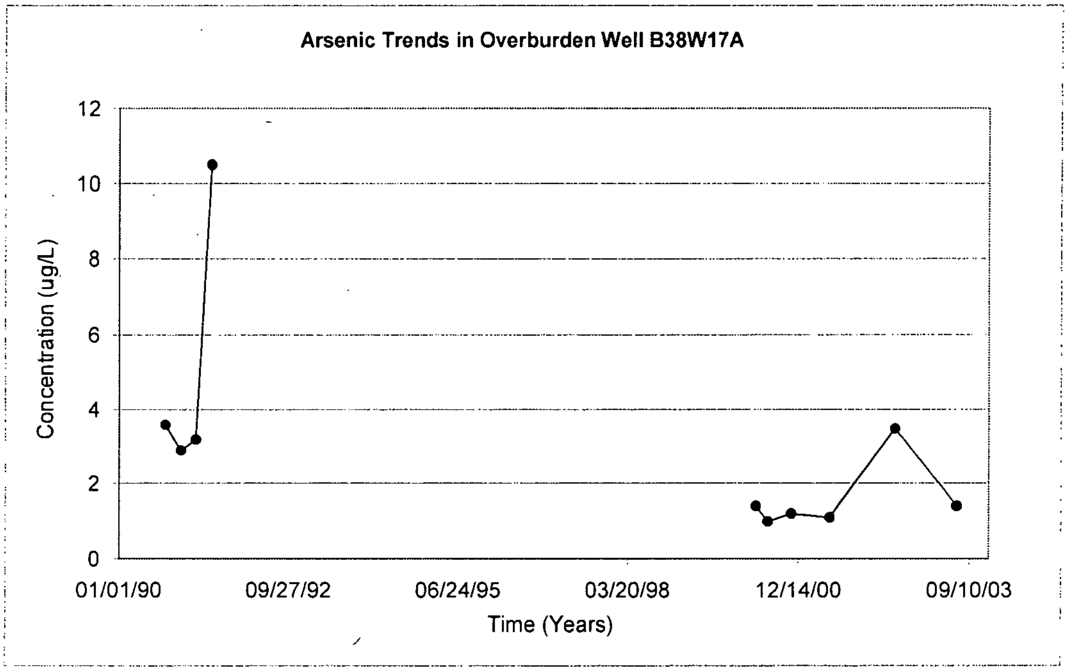

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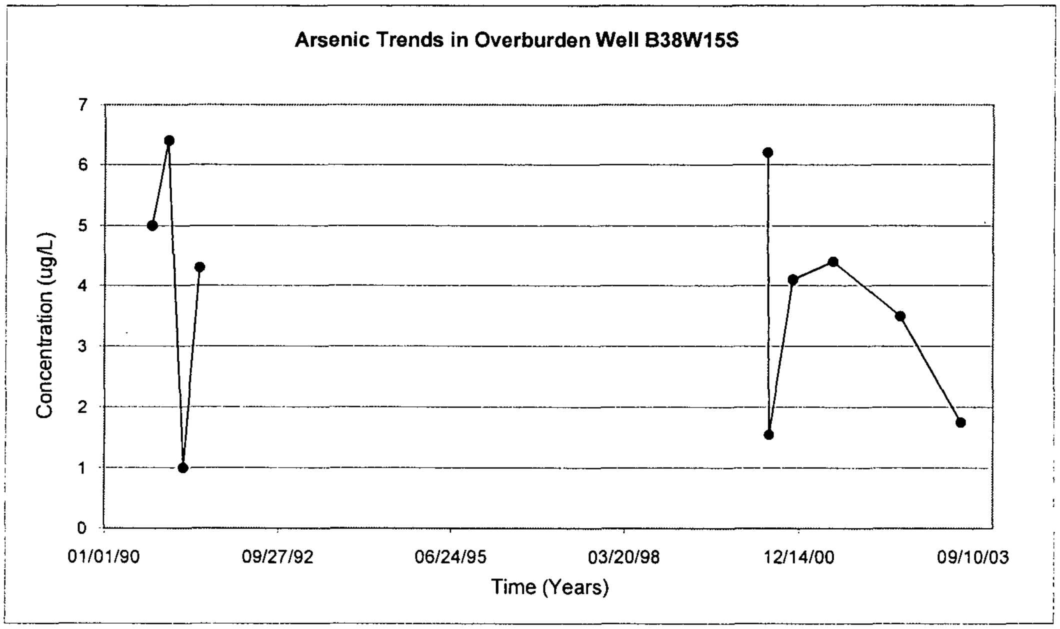

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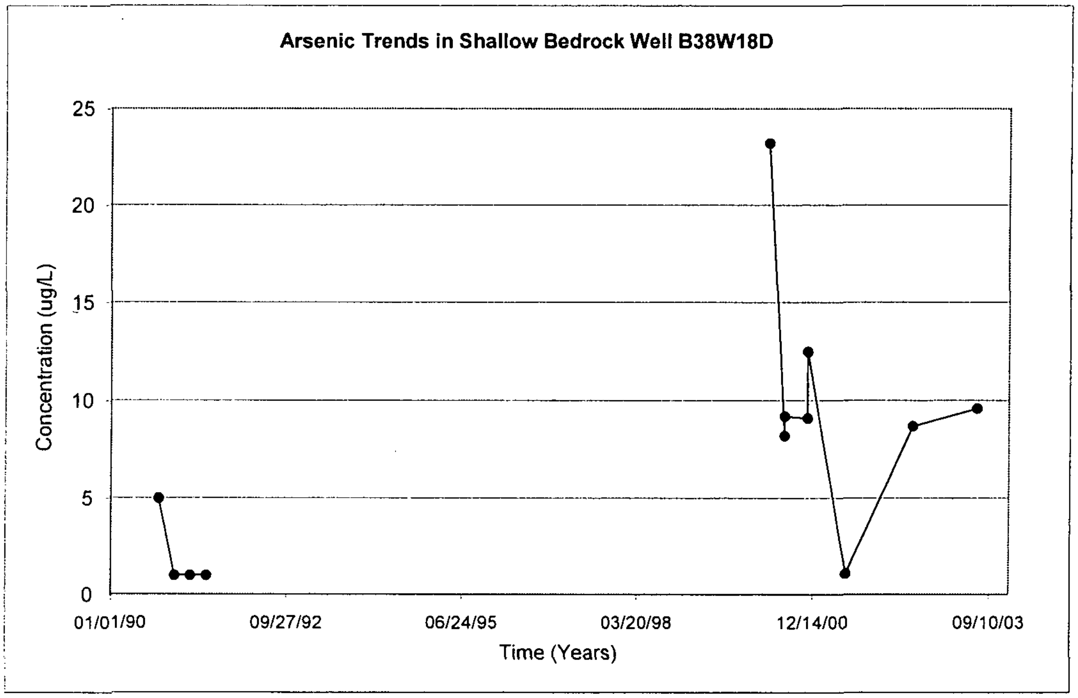

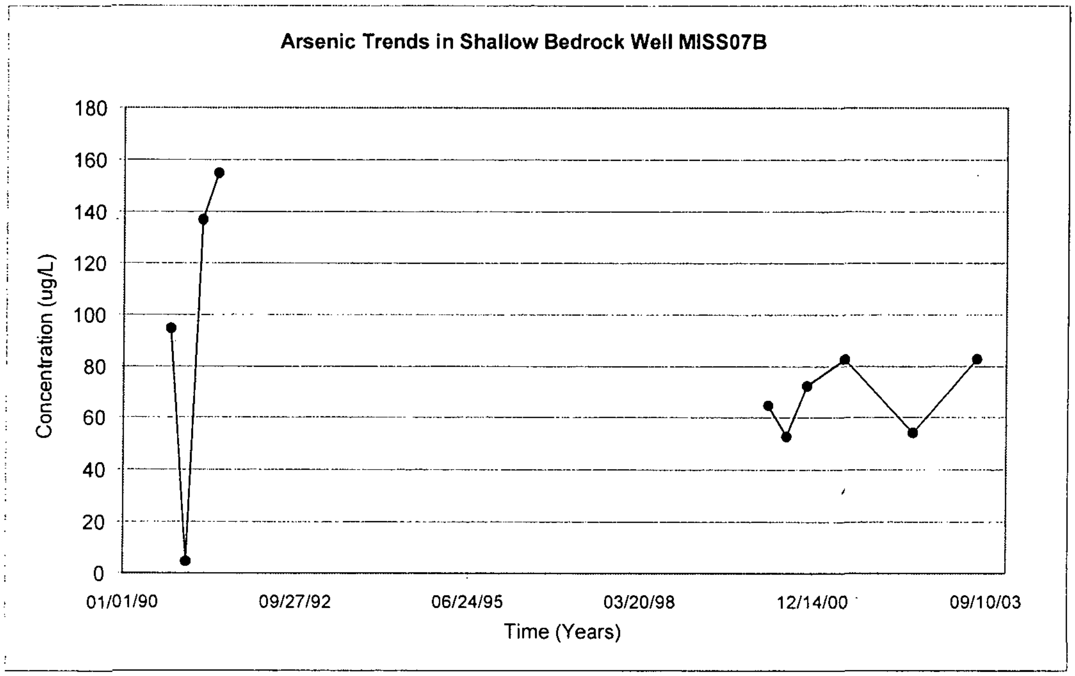

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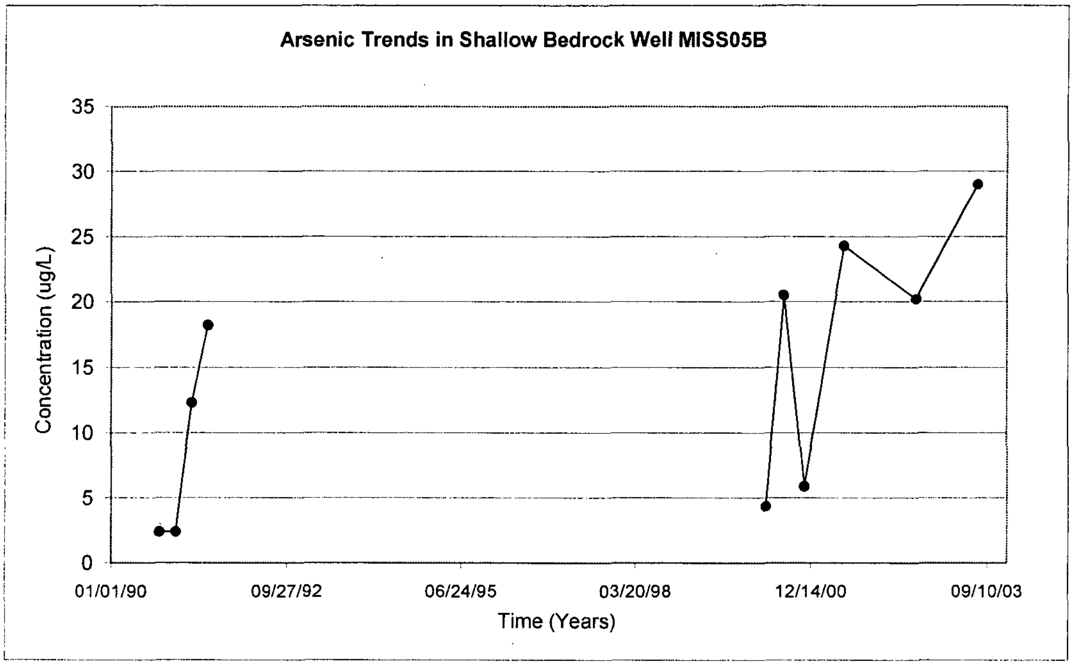

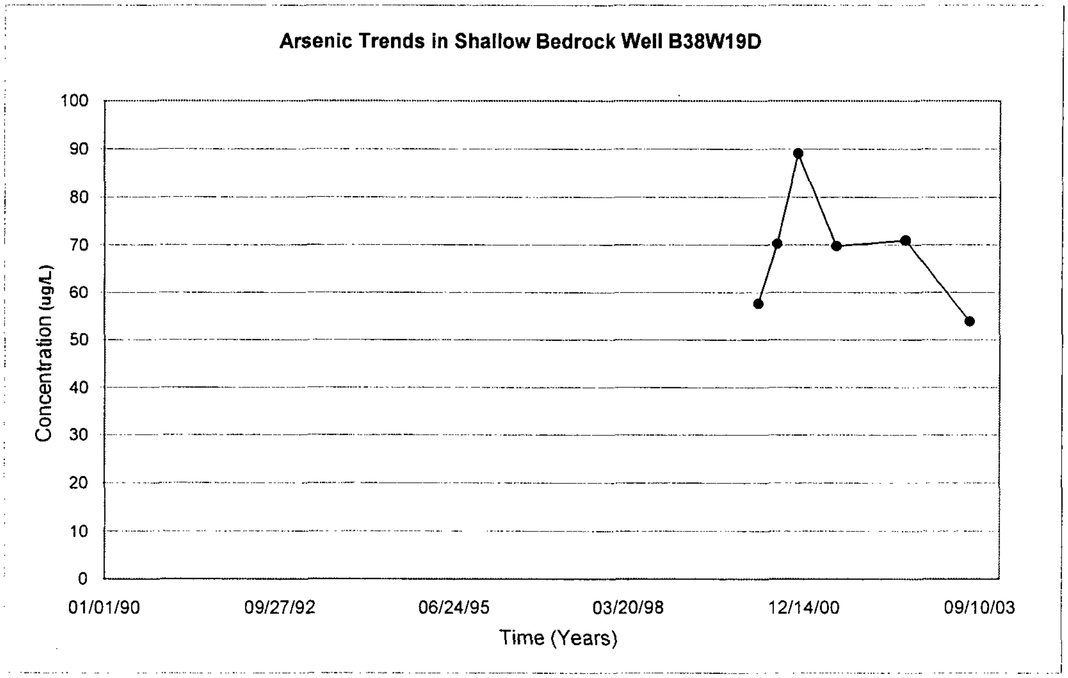

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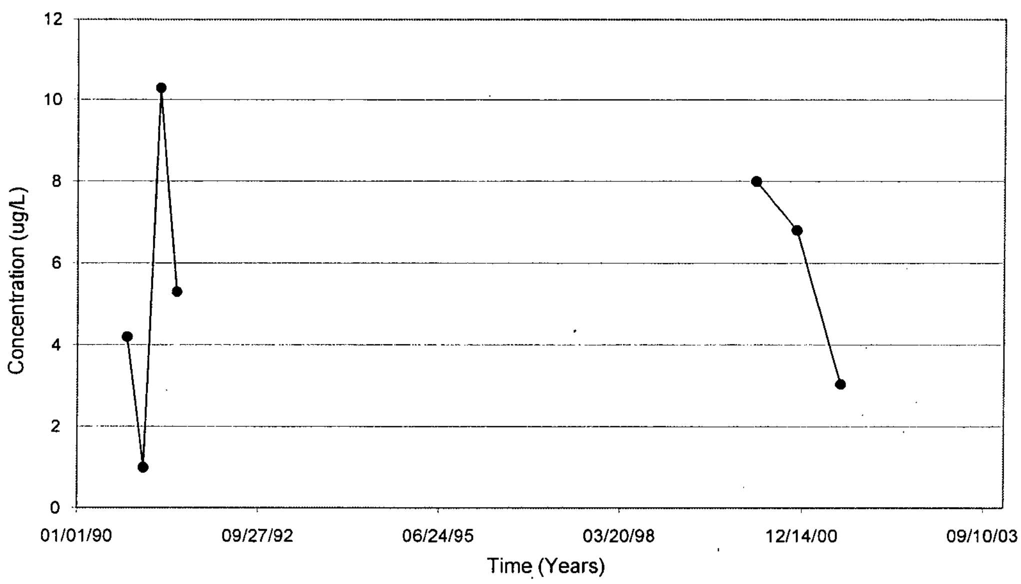
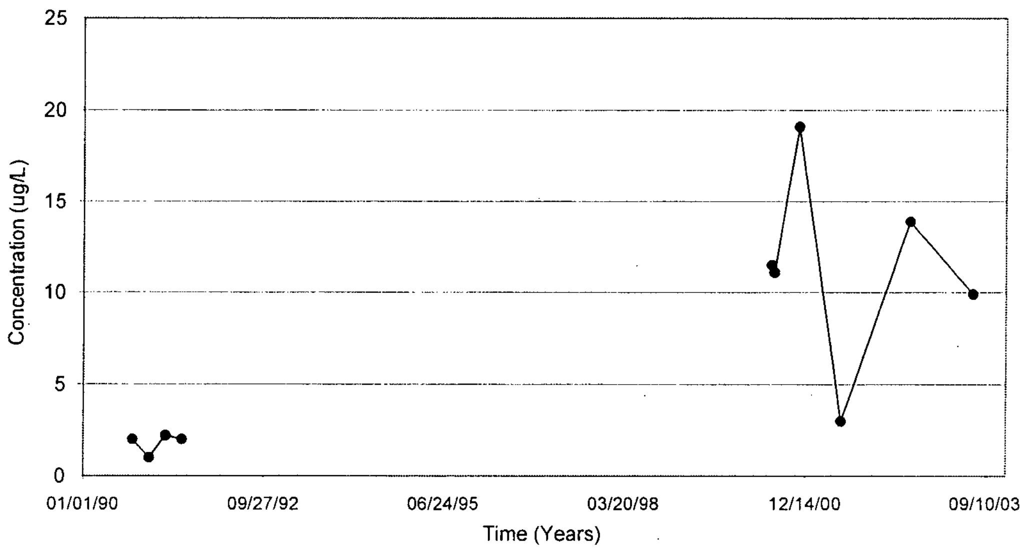
Arsenic Trends in Shallow Bedrock Well MISS03B



Arsenic Trends in Shallow Bedrock Well B38W15D



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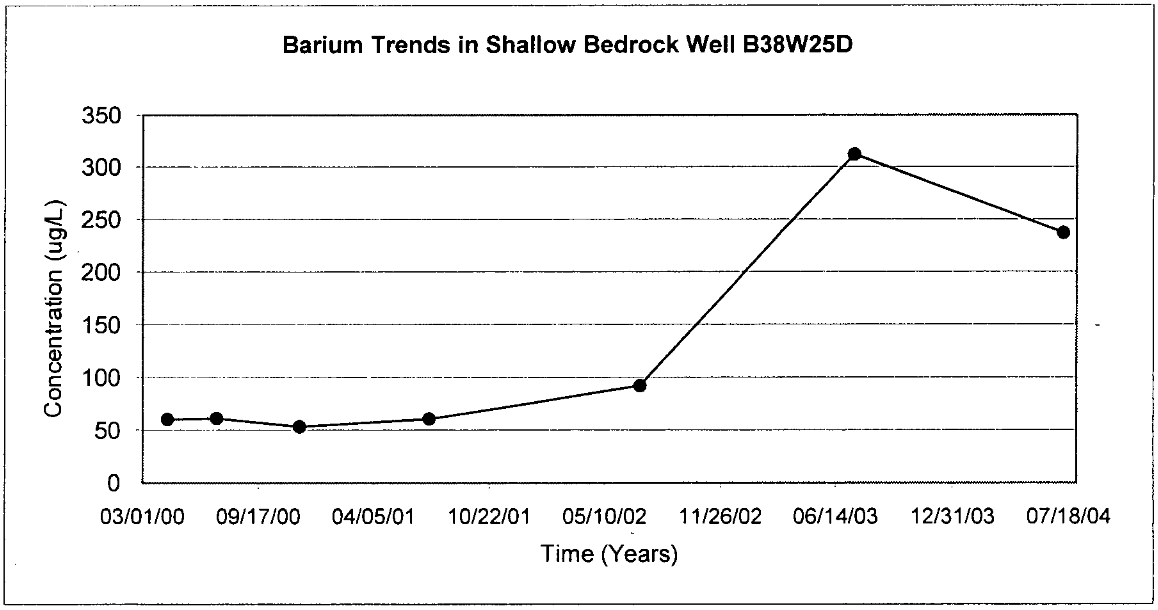

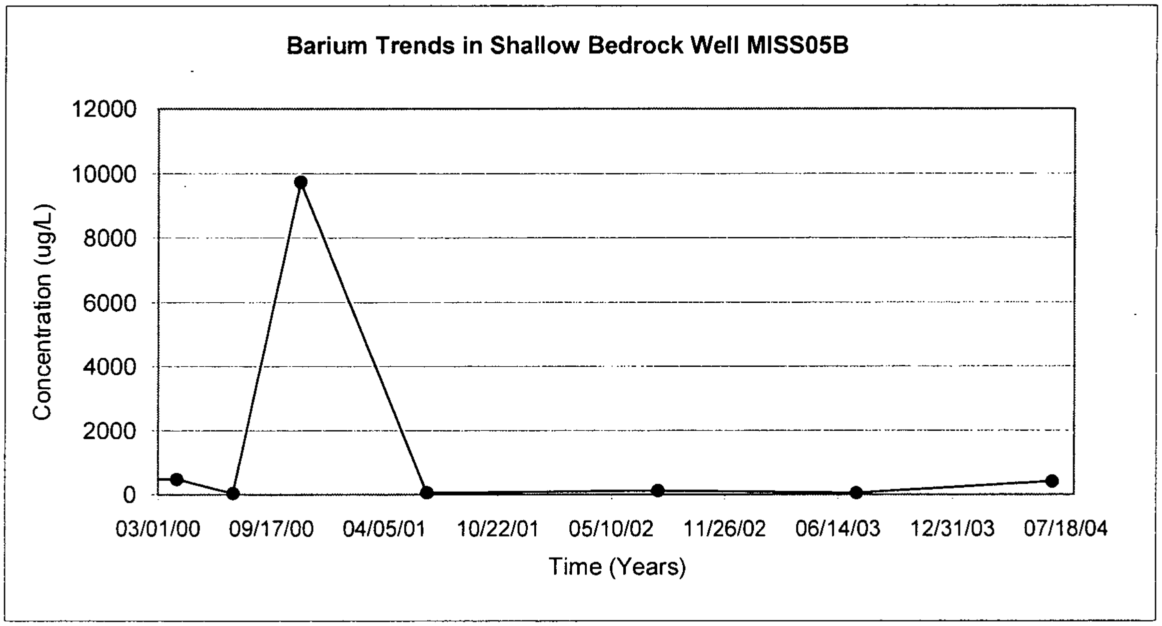

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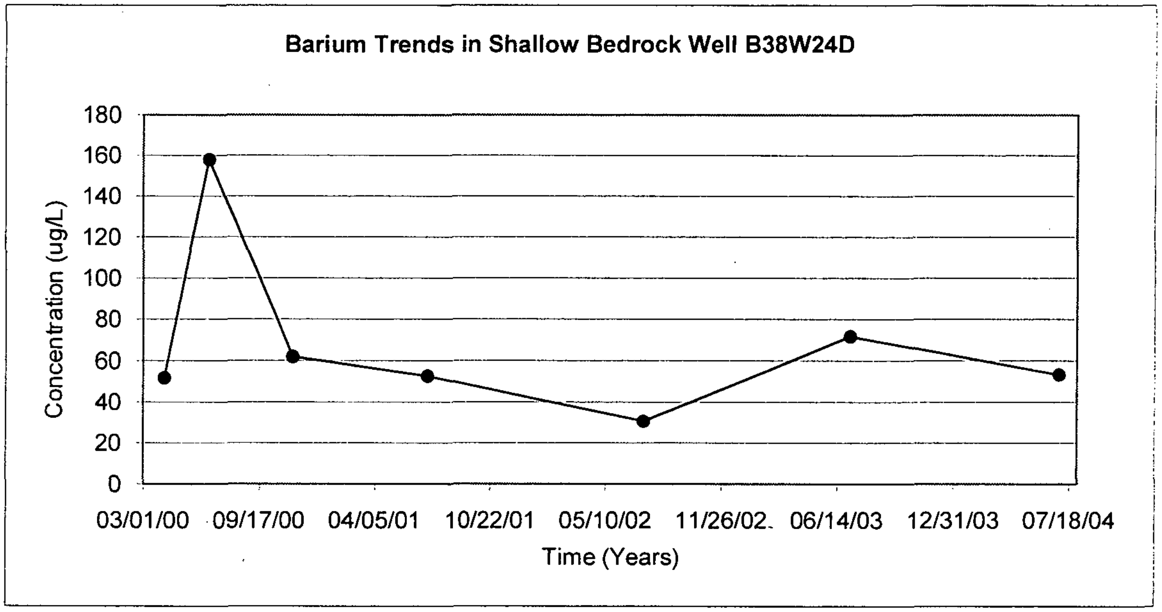

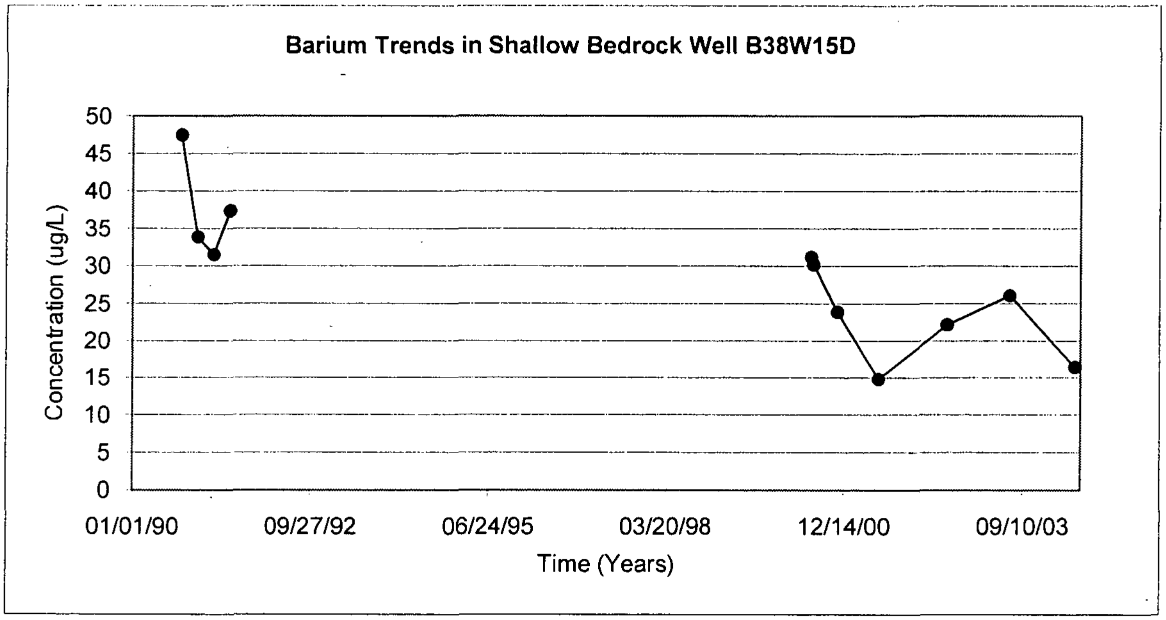

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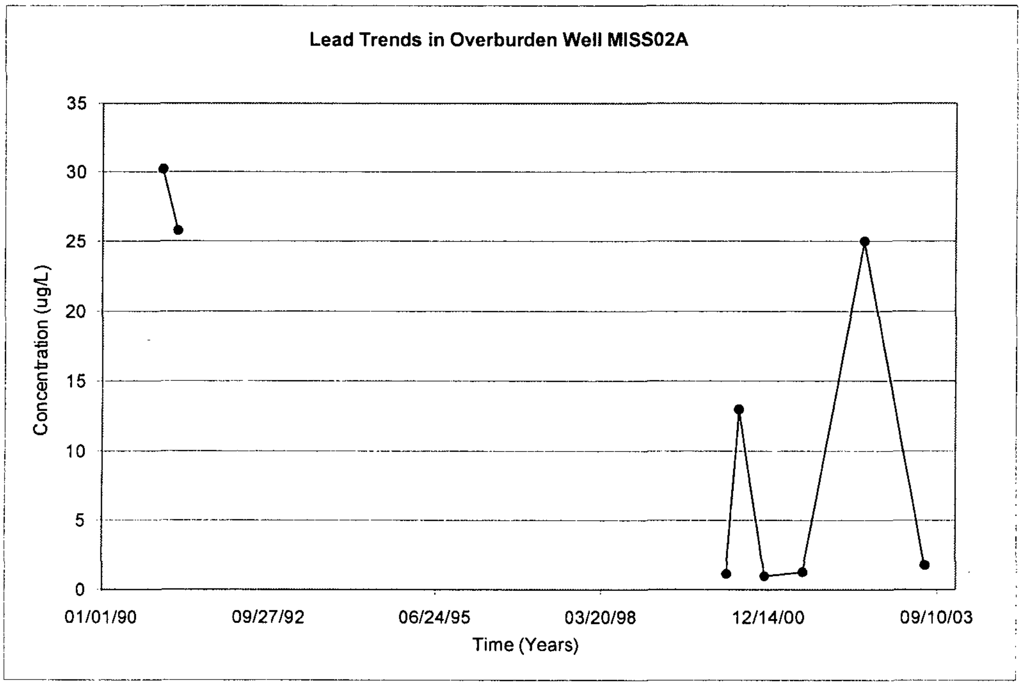

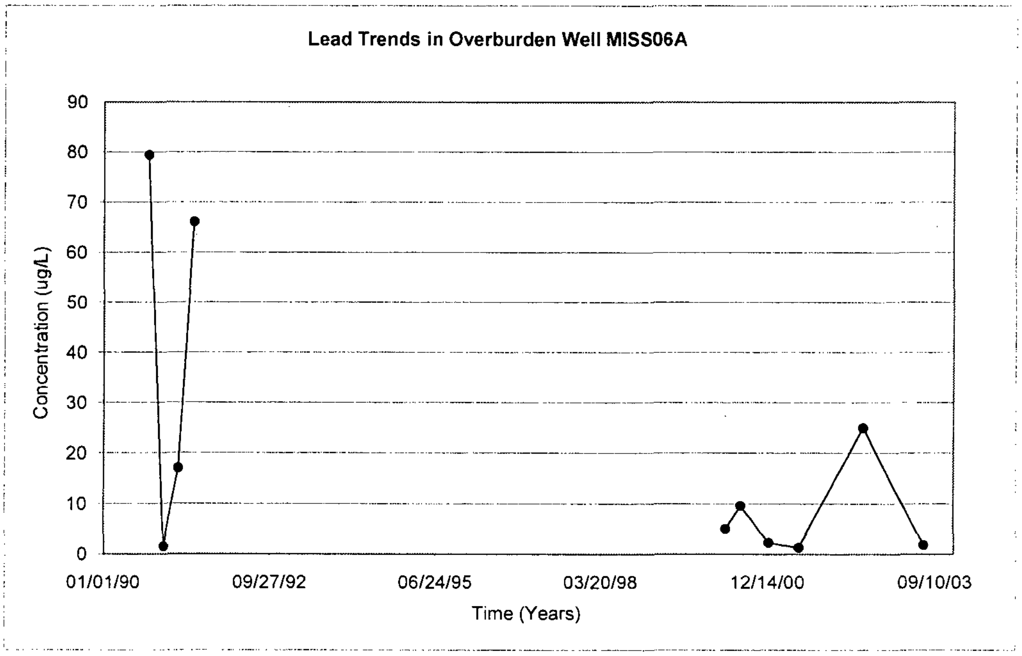

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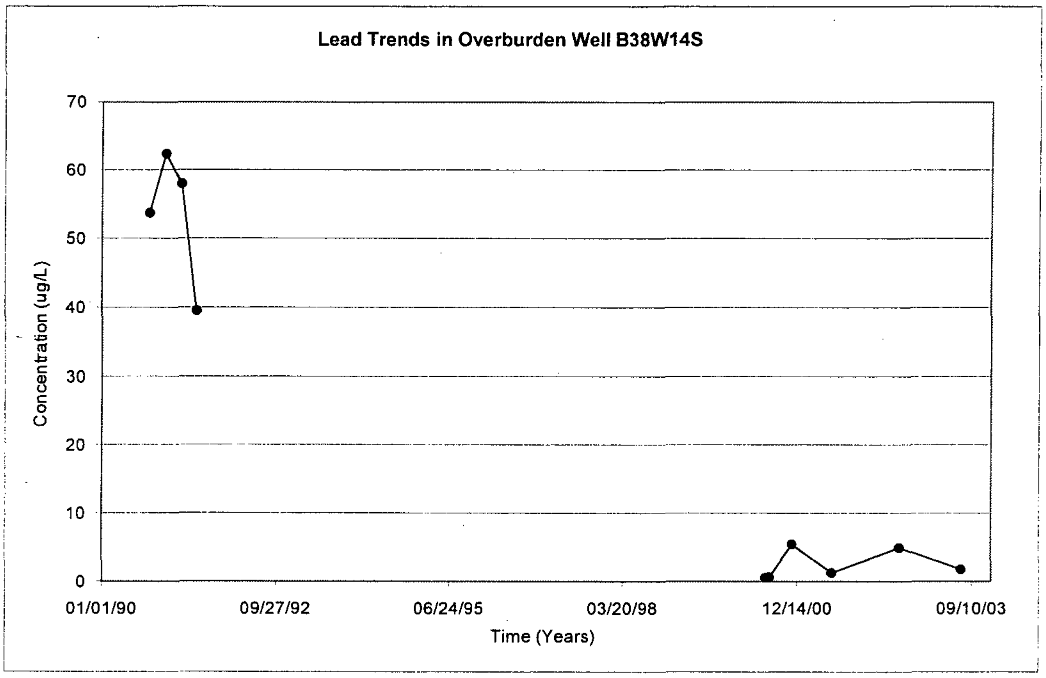

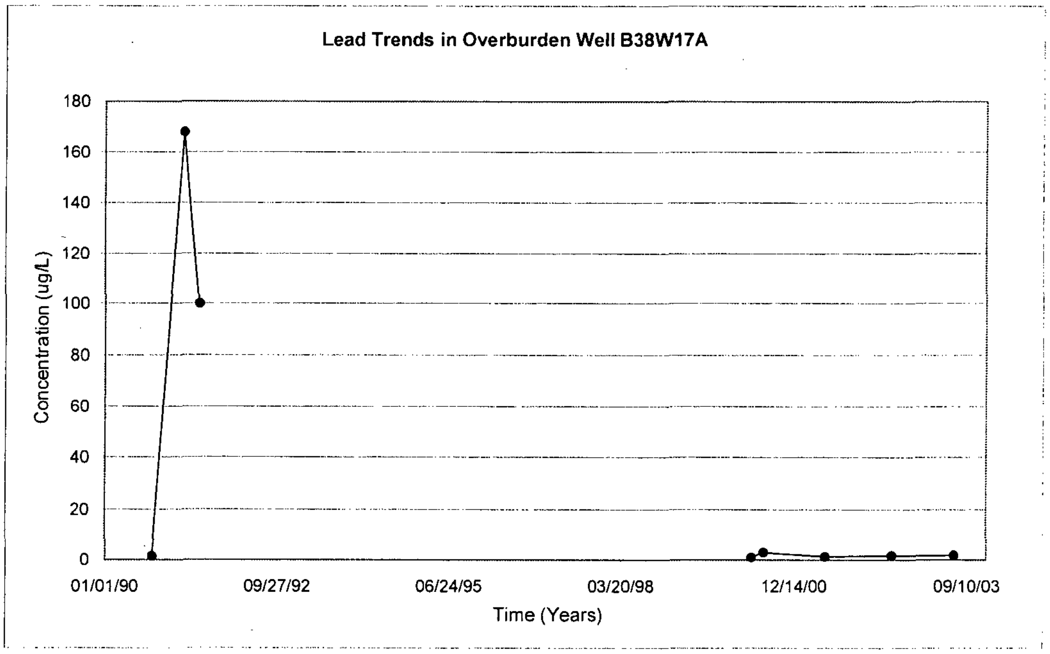

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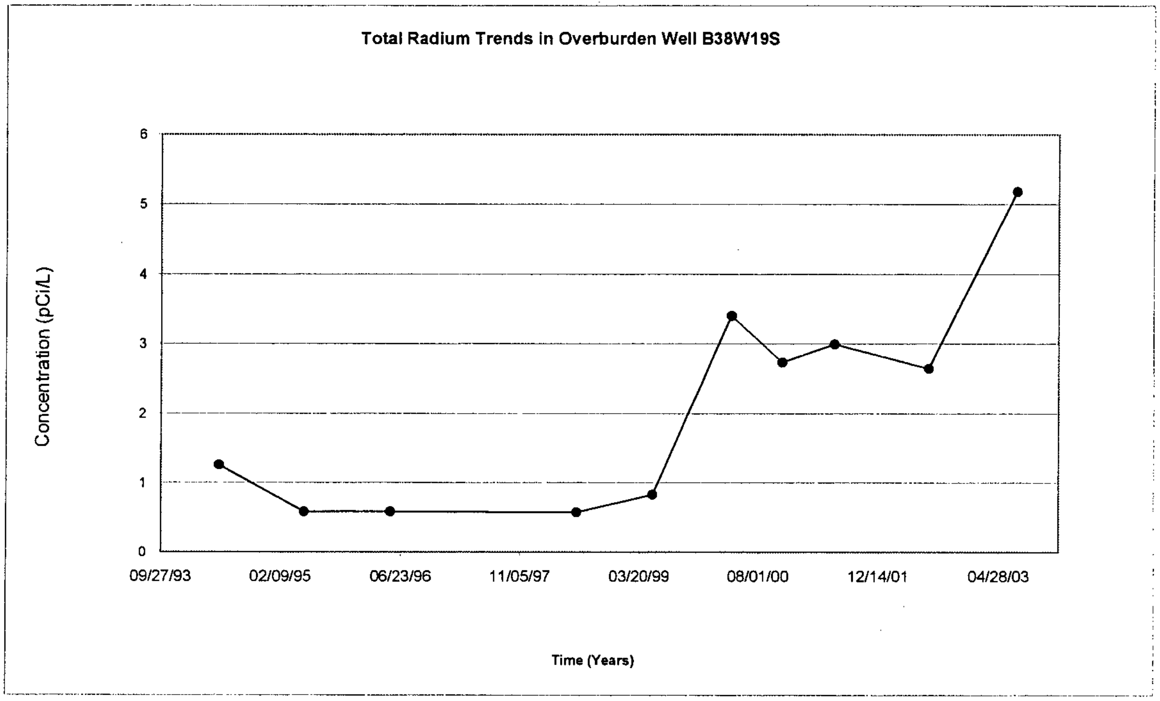

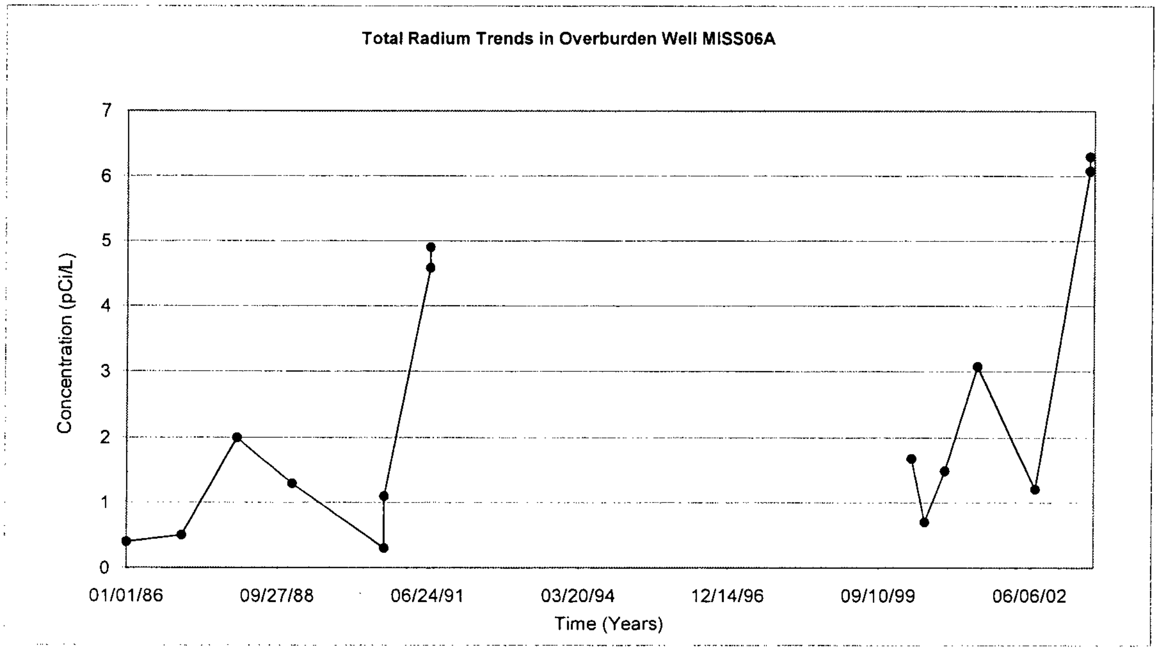

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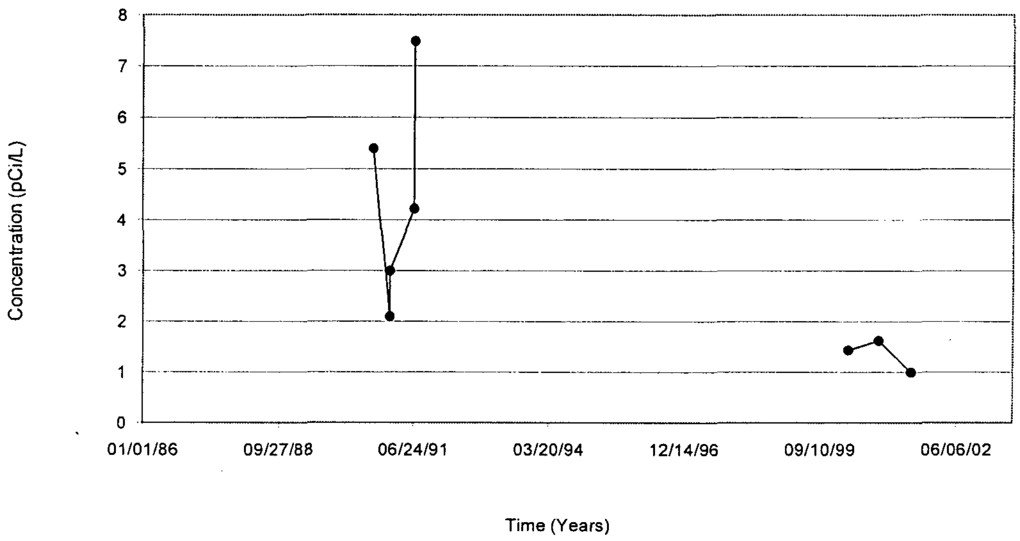

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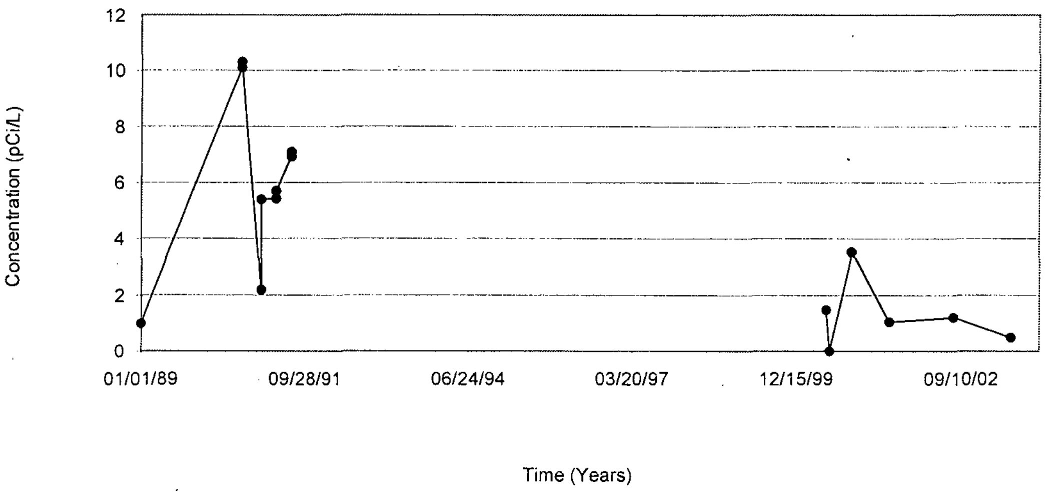

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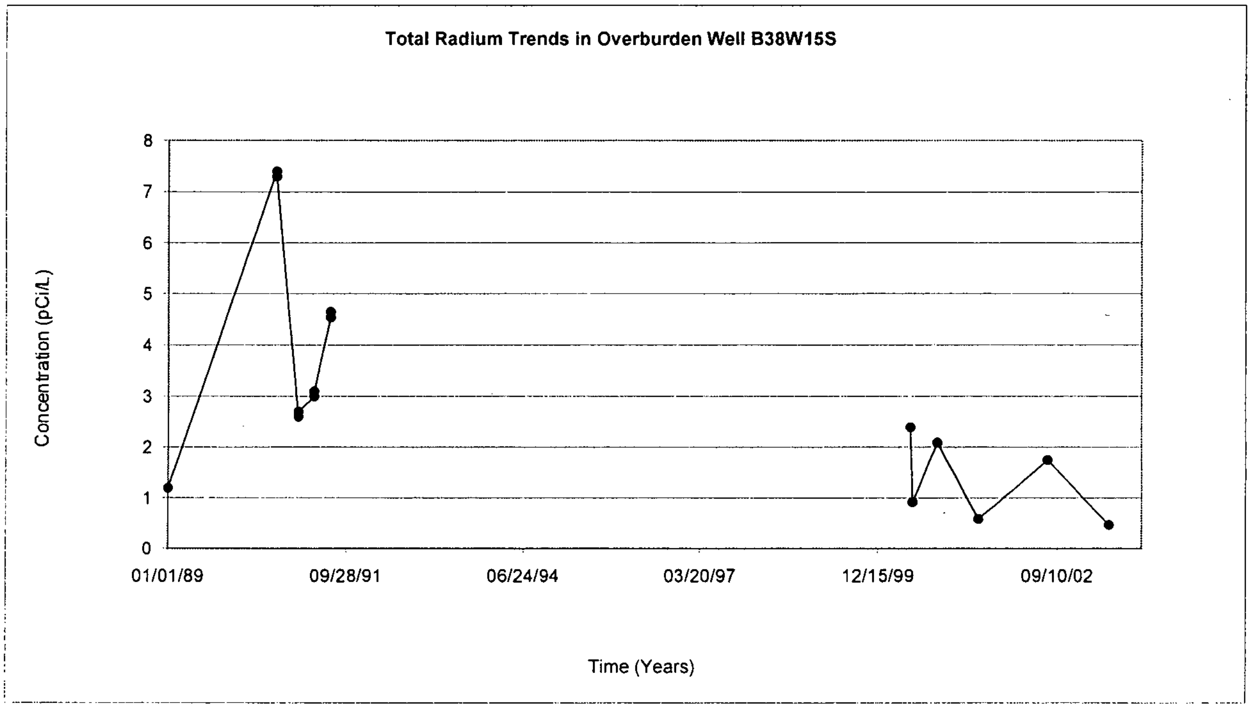

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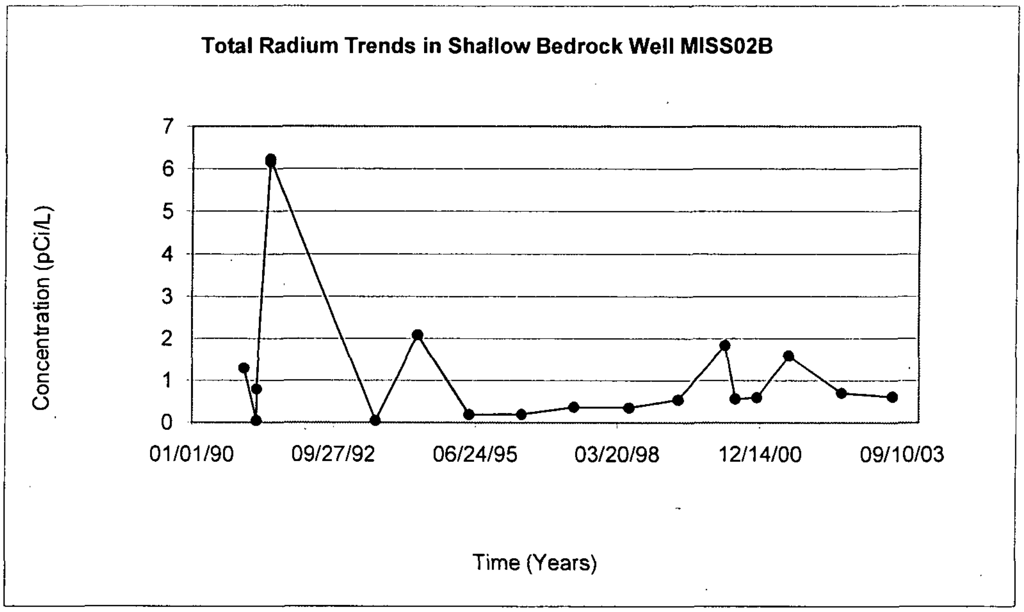

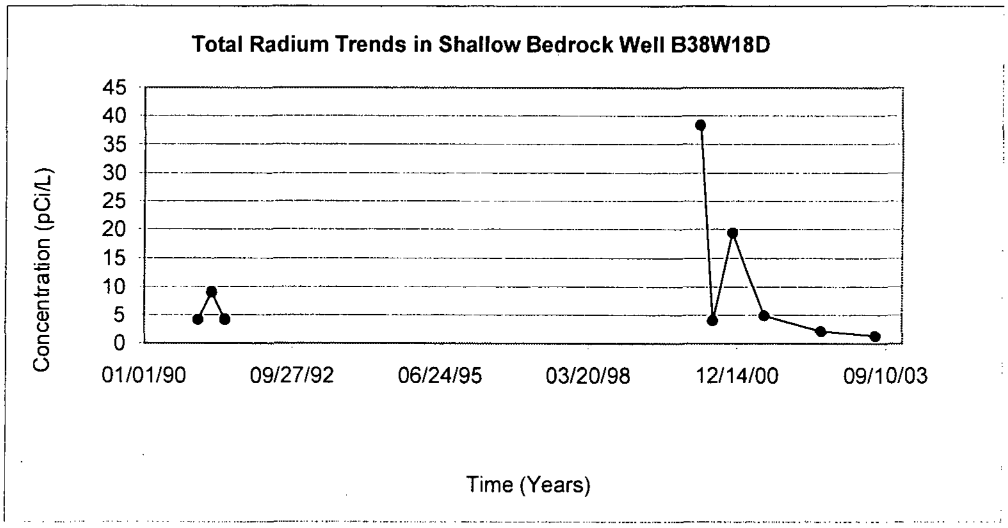

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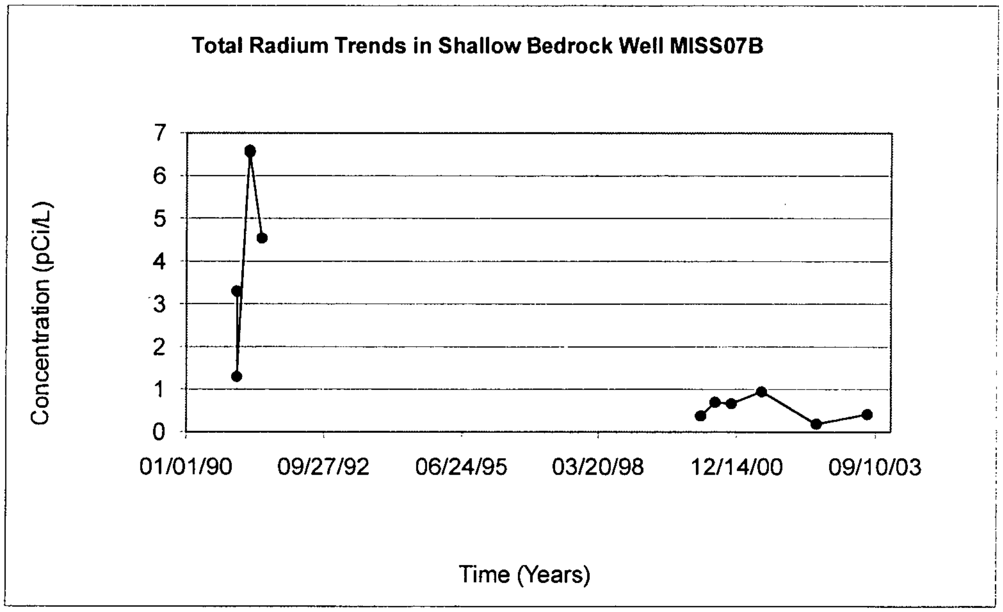

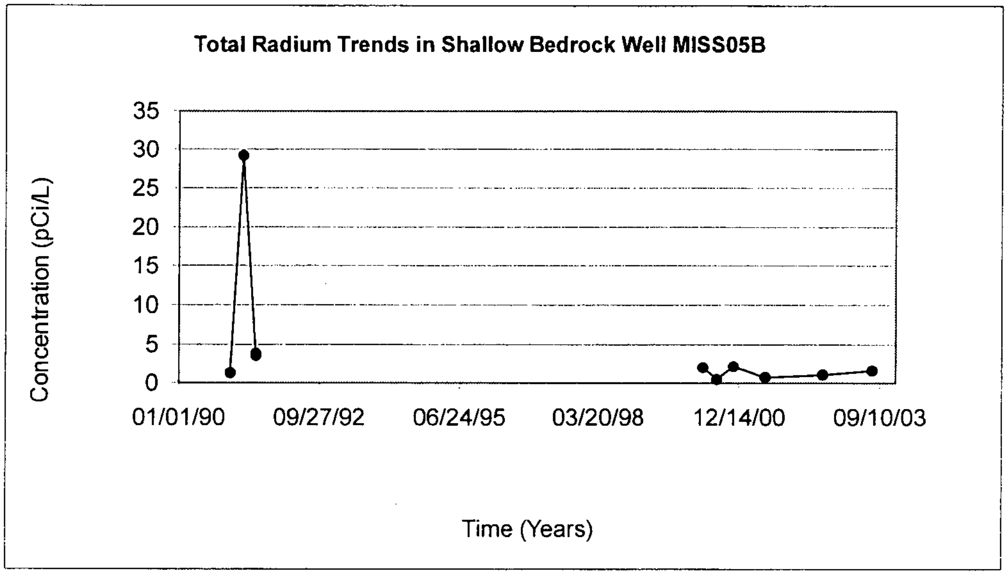

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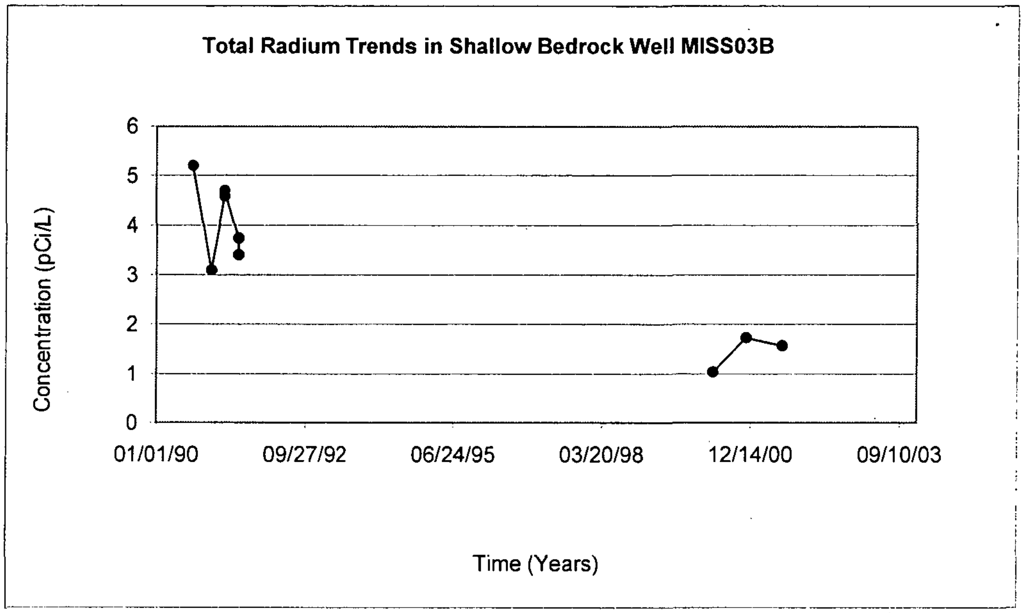

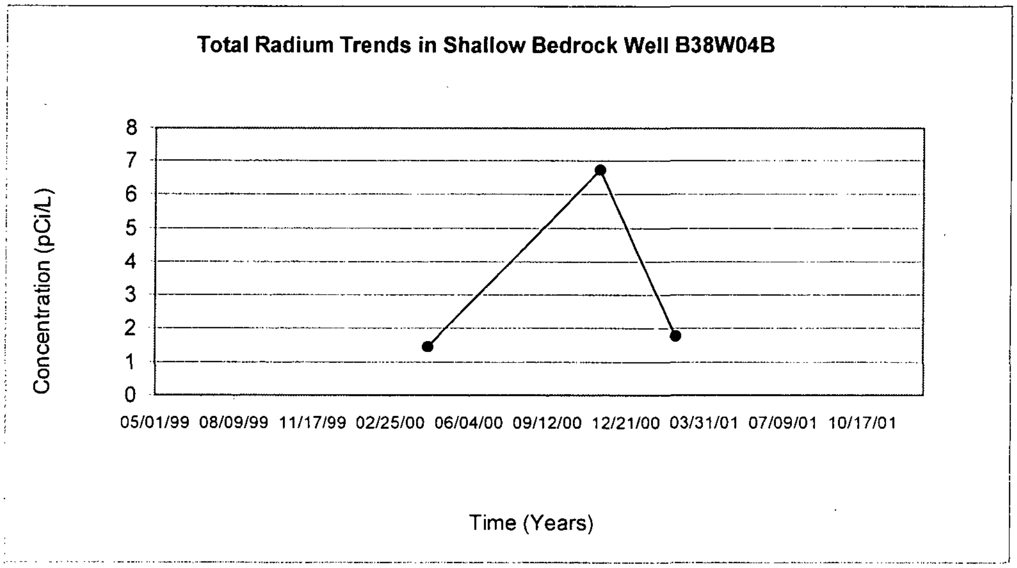

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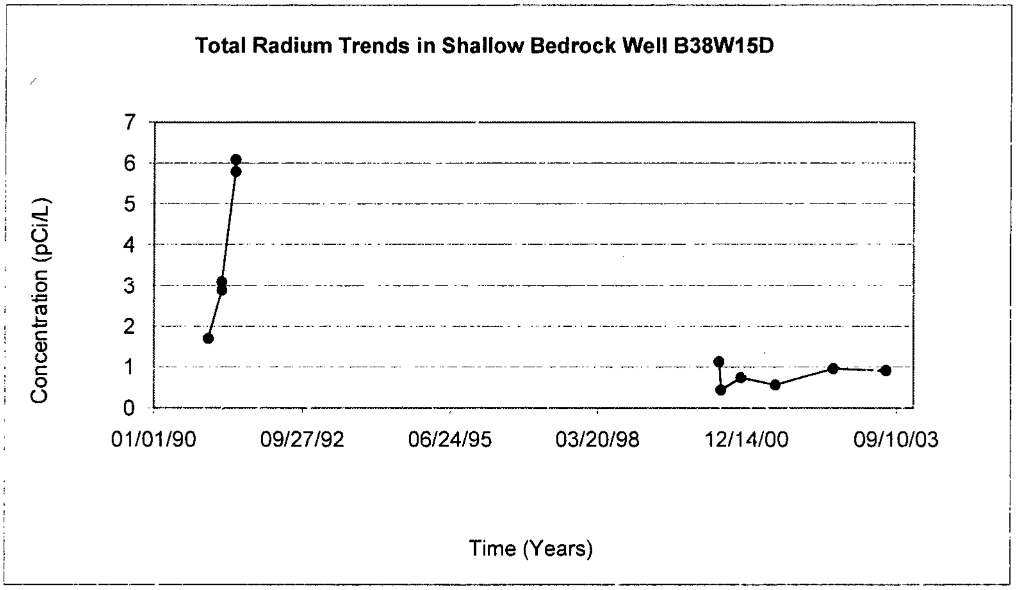

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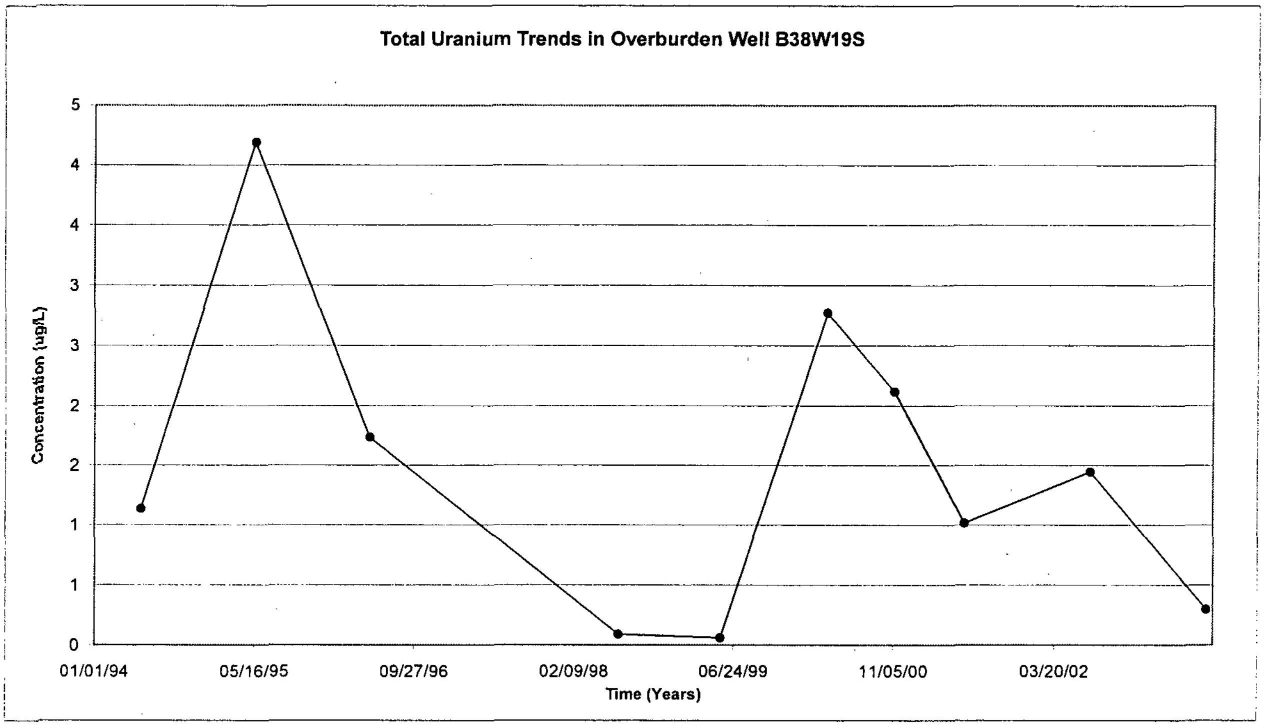

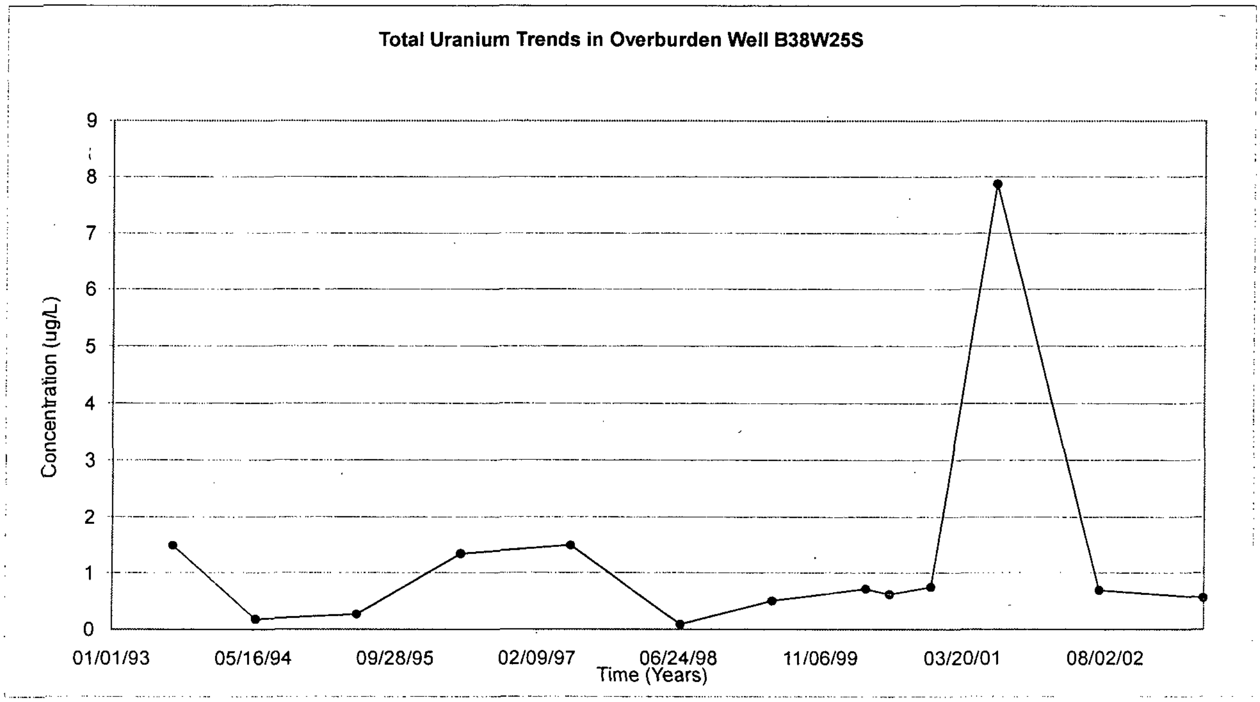

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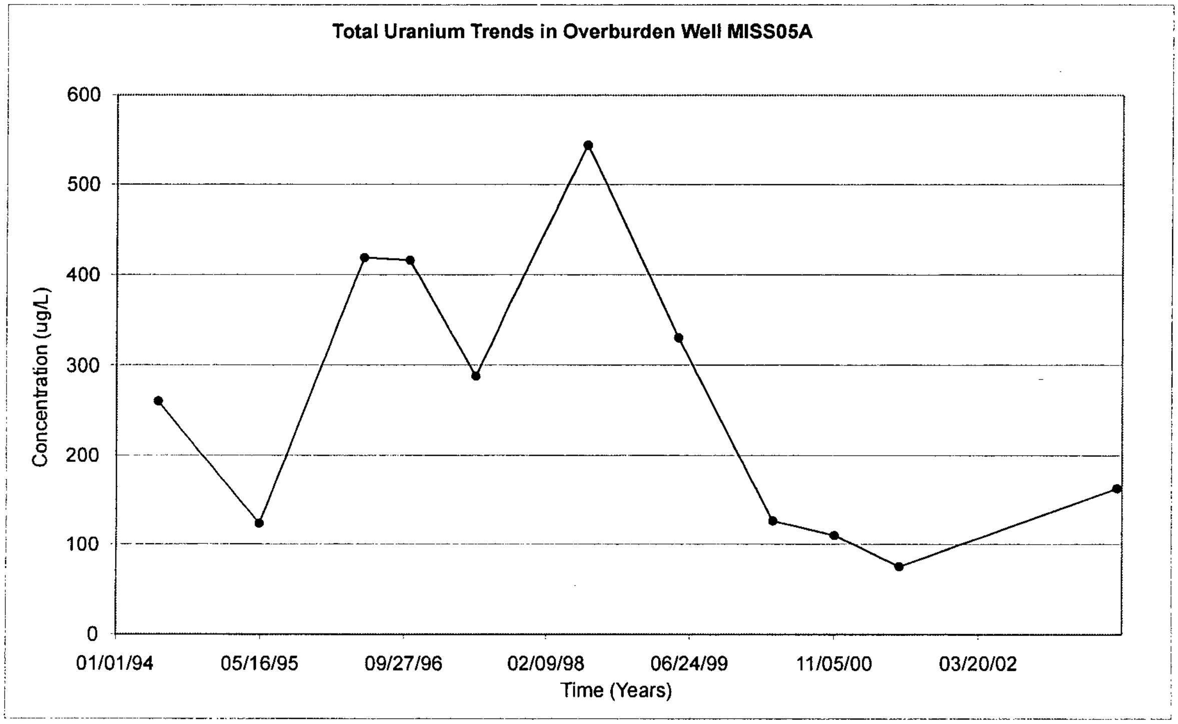

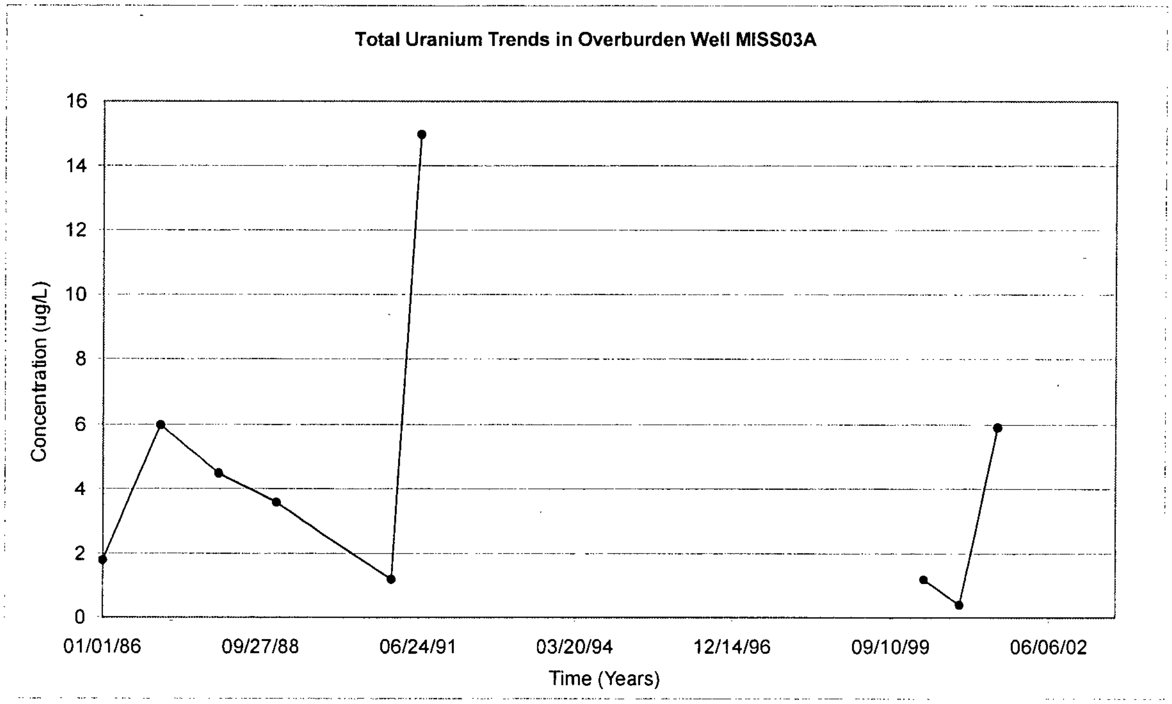

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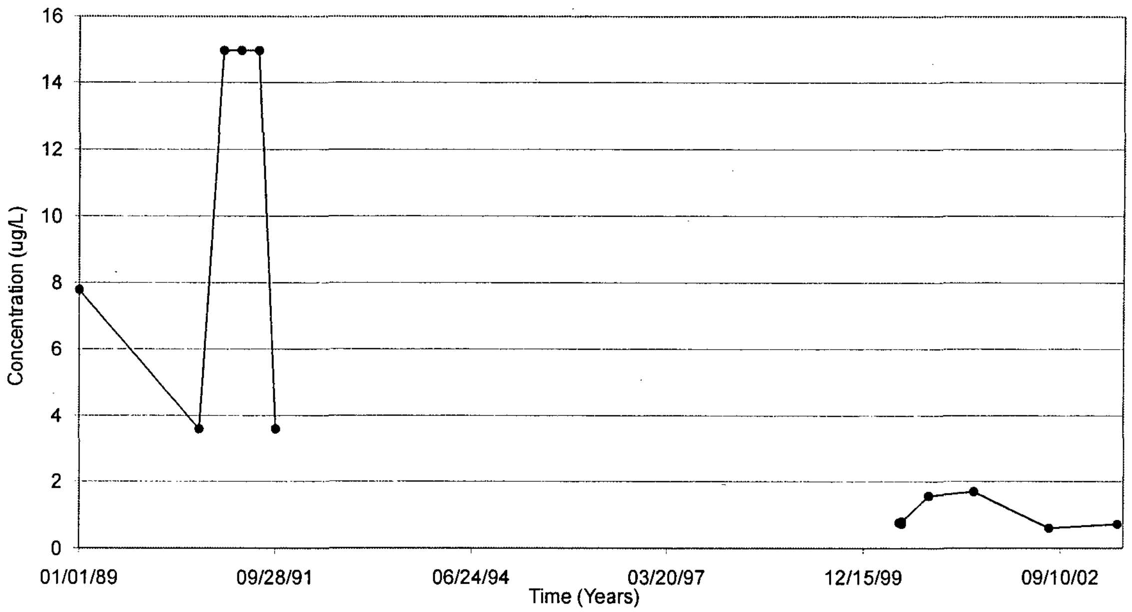
Total Uranium Trends in Overburden Well B38W15S



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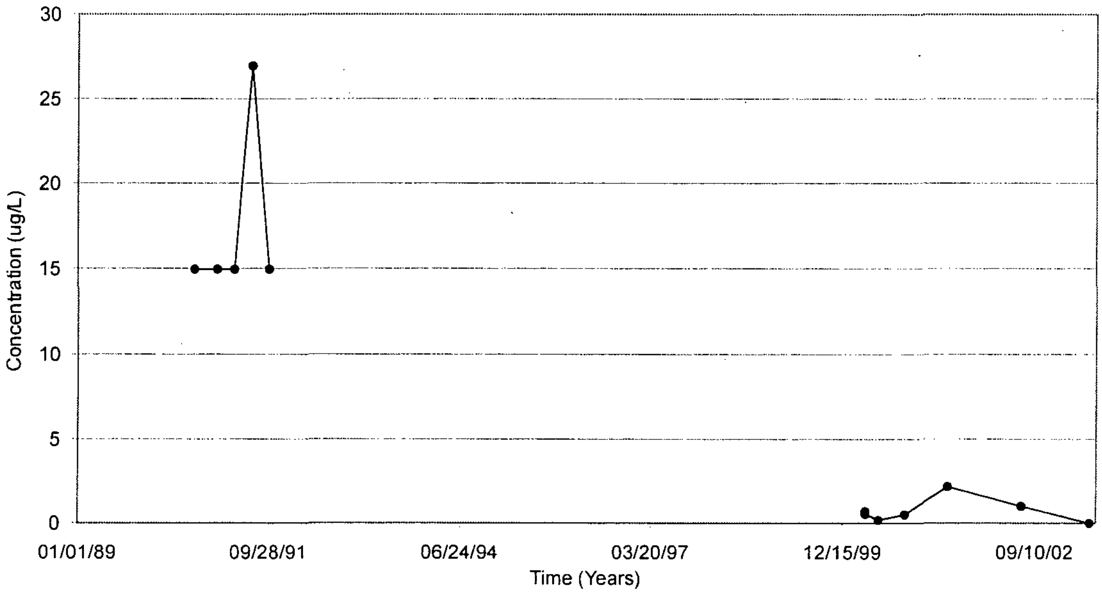

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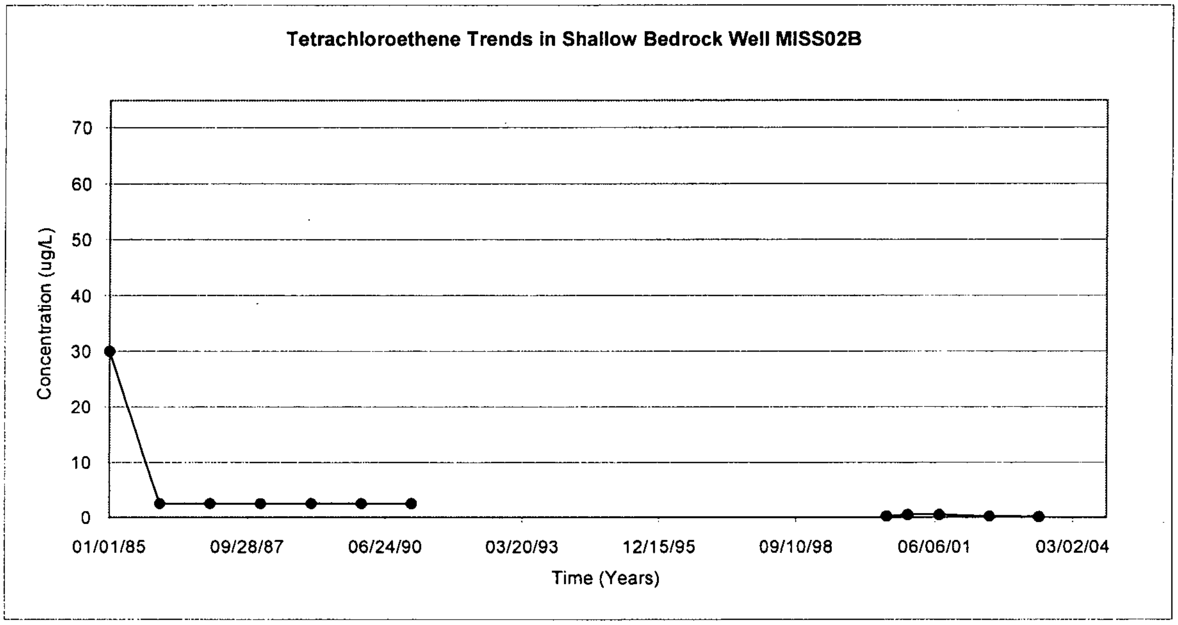

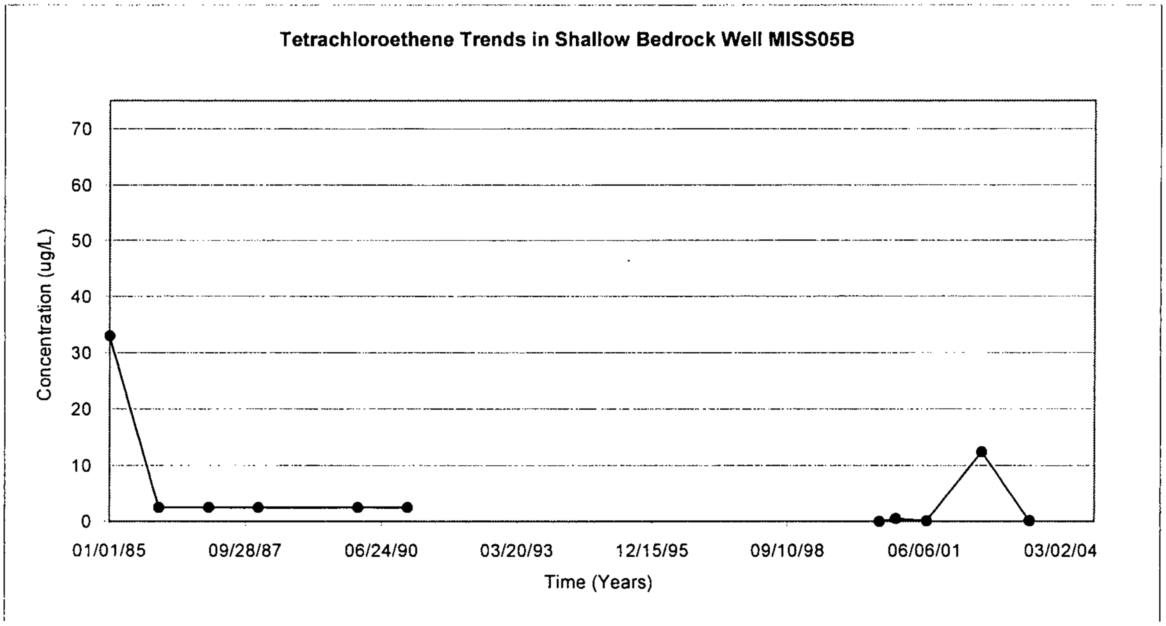

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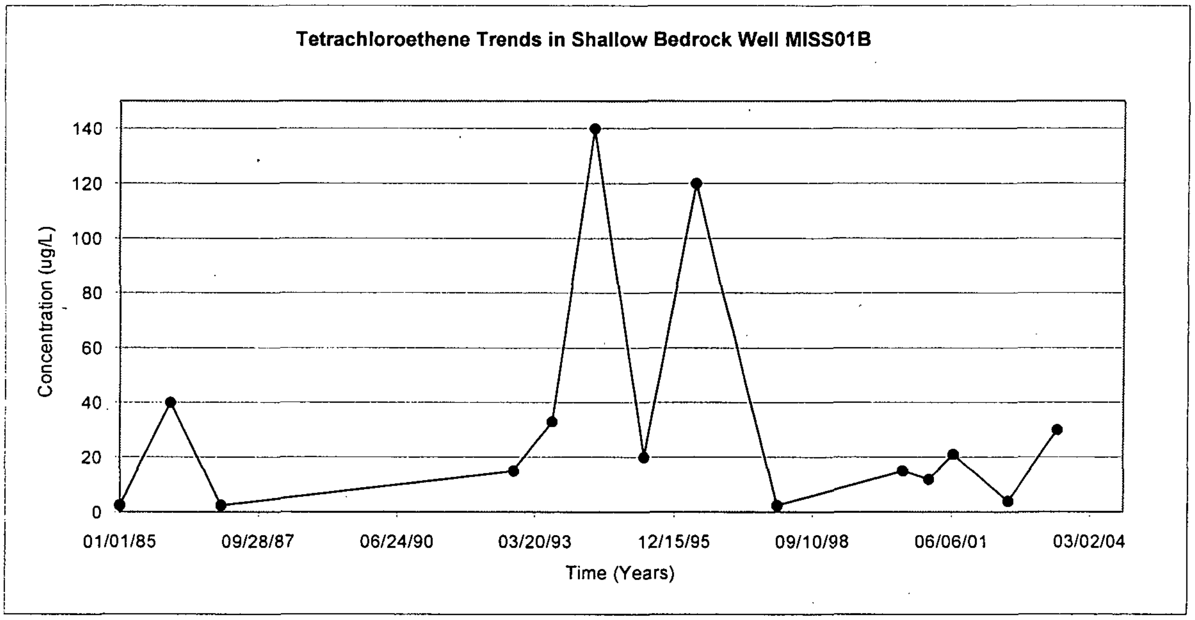

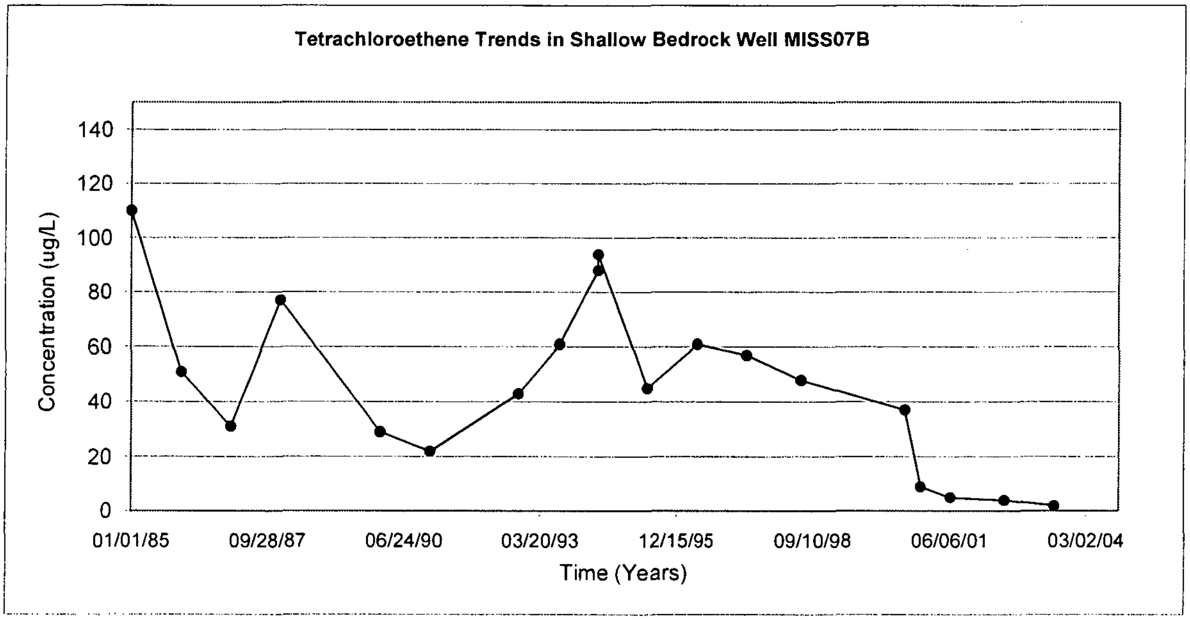

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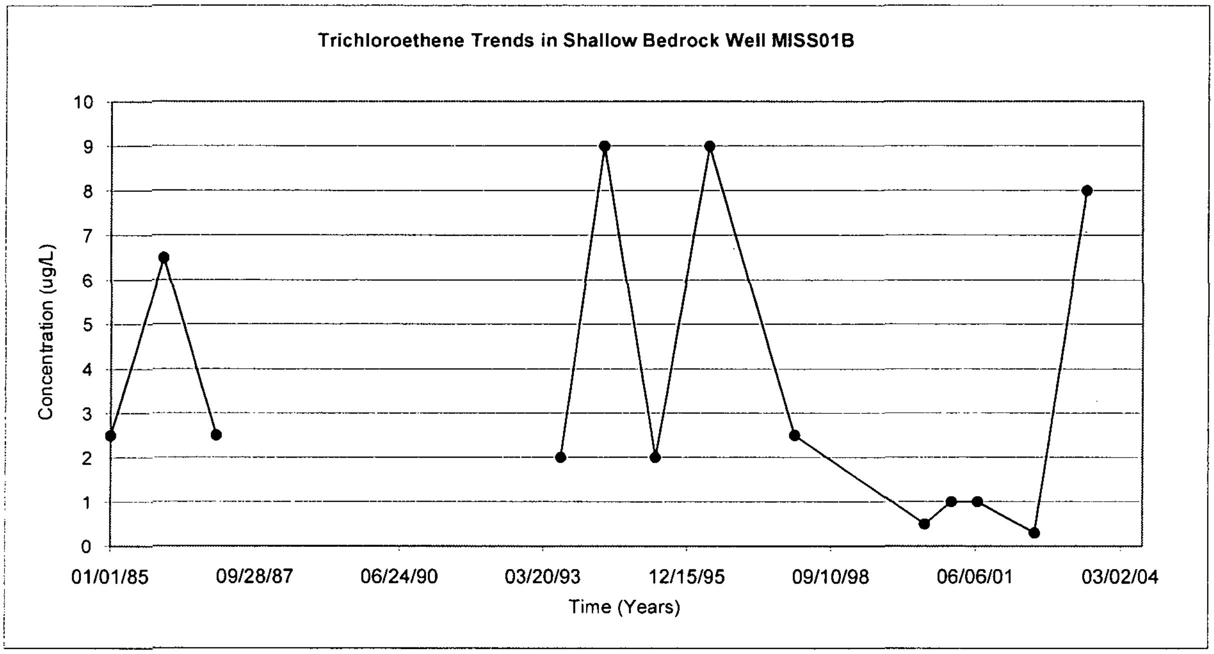

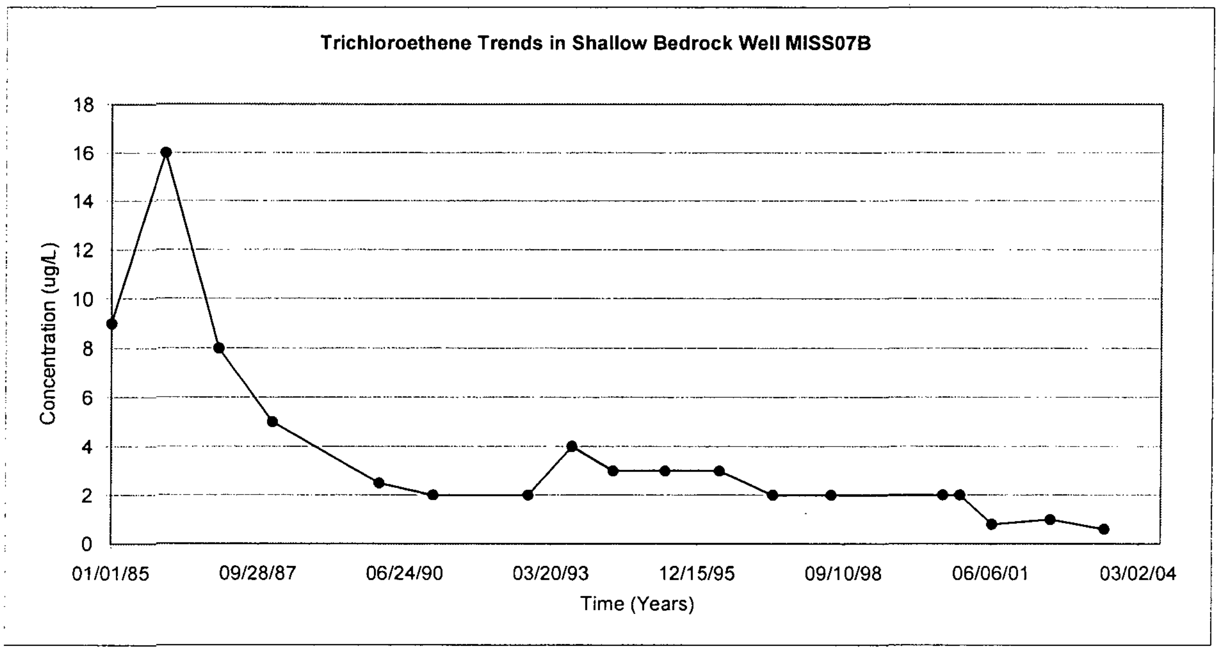

APPENDIX B

CONTAMINANT TRANSPORT PARAMETERS AND IN-SITU REMEDIATION METHODS FOR SELECTED METALS, RADIONUCLIDES, AND ORGANIC COMPOUNDS IN GROUNDWATER (VOLUME 1)

APPENDIX B

Contaminant Transport Parameters and In-Situ Remediation Methods for Selected Metals, Radionuclides, and Organic Compounds in Groundwater at FUSRAP Maywood Superfund Site

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1.0 Introduction

This appendix provides contaminant transport parameters and discusses potential remediation methods for selected metals, radionuclides, and organic compounds in groundwater at the FUSRAP Maywood Superfund Site (FMSS). The purpose and objectives of this appendix are to support the detailed analysis of alternatives and contaminant transport modeling for the Groundwater Feasibility Study (GWFS) for the Maywood Interim Storage Site (MISS). The site-specific data used to support the derivation of transport parameters and discussions of potential in-situ remediation methods are from the *Final Groundwater Remedial Investigation Report (GWRI), Revision 4* (July 2005). An initial set of transport parameters was provided in Chapter 6 of the GWRI for some of the constituents. This appendix provides additional parameters that are required for the GWFS evaluations.

Transport parameters that are provided include adsorption coefficients for selected metals, radionuclides, and organic compounds (**Section 2.0**). Also provided are discussions of the geochemical behavior, in-situ treatment methods, and treatment half-lives of selected metals, organic compounds, and radionuclides (**Section 3.0**); half-lives and activity-to-mass conversion factors for selected radionuclides (**Section 4.0**); and an evaluation of the sources of gross alpha and gross beta activities detected in groundwater samples (**Section 5.0**).

2.0 Adsorption Coefficients

This section provides a compilation of adsorption coefficients (K_d) for use in contaminant transport modeling at the FMSS. Coefficients are provided for the metals barium, beryllium, iron, lead, and thallium; and the organic compounds 2-chlorotoluene, methylene chloride, tetrachloroethene (PCE), toluene, trichloroethene (TCE), vinyl chloride (VC), and total xylenes. Literature K_d values for benzene, thorium, uranium, radium, arsenic, chromium, and lithium have already been presented in the GWRI. Those values were reviewed as part of this evaluation, and were found to be acceptable.

Table 1 provides the recommended K_d values for the metals. **Table 2** provides the recommended values of the organic carbon adsorption coefficients (K_{oc}) for the organic compounds, and **Table 3** provides the corresponding K_d values for the organic compounds.

2.1 Metals

This section provides adsorption coefficients for barium, beryllium, iron, lead, and thallium.

2.1.1 Barium

Estimates of barium K_d values are provided in the *Soil Screening Guidance: Technical Background Document* (EPA,1996) as a function of pH. The K_d value at a pH of 6.8, which is close to the mean of site pH values, is 41 mL/g. An experimental study of barium sorption on glacial till at a site in Ohio (IT Corporation, 1993) yielded a range of K_d values from 66 to 128 mL/g. A realistic value to use for modeling purposes would be the geometric mean of these three values, which equals 70 mL/g.

2.1.2 Beryllium

Estimates of beryllium K_ds are provided in the *Soil Screening Guidance: Technical Background Document* (EPA,1996) as a function of pH. The K_d value at a pH of 6.8, which is close to the mean of site pH values, is 790 mL/g. This value is quite close to a K_d of 800 mL/g provided by Sheppard and Thibault (1990) for loam soil. A realistic value to use for modeling purposes would be 790 mL/g.

2.1.3 Iron

Iron can exist in the divalent [ferrous, or Fe (II)] or trivalent [ferric, or Fe (III)] forms in the sedimentary environment. These two valence states have very different behavior. Dissolved iron concentrations in groundwater are controlled mostly by dissolution–precipitation reactions rather than adsorption–desorption reactions. The oxidation-reduction (redox) state of the aquifer environment is the key parameter controlling iron concentrations in groundwater. Under oxidizing conditions where Fe (III) is stable, the solubility of iron is well below the standard analytical reporting limits. Detectable concentrations of iron in oxic groundwater samples are usually due to the presence of iron-bearing suspended particulates rather than dissolved iron. If the redox potential is below a critical value (which is a function of pH), then Fe (II) becomes stable. The solubility of Fe (II) can be as high as several tens of milligrams per Liter (mg/L).

The addition of organic contaminants such as fuel or solvents to an aquifer usually results in the establishment of locally reducing conditions within the contaminant plume. Within the reducing zone, naturally occurring iron and manganese oxides that are part of the aquifer matrix become soluble and dissolve, leading to elevated iron and manganese concentrations. Trace metals such as arsenic, nickel, and vanadium, which naturally adsorb on iron oxides, can also be released to the groundwater as the oxide minerals dissolve. Iron, manganese, and trace element concentrations will remain elevated within this zone, as along as reducing conditions prevail. Iron is most realistically simulated in a transport model by considering solubility as a function of local redox conditions rather than treating it as a sorbing solute. Therefore, adsorption coefficients for iron were not evaluated in this appendix.

2.1.4 Lead

A detailed literature review of lead adsorption behavior was performed by EPA (1999). The review concluded that the adsorption of lead in the environment is influenced by a number of factors such as the type and properties of adsorbing substrate, pH, the equilibrium concentrations of dissolved lead, and the type and concentrations of other competing cations and complex-forming inorganic and organic ligands. These data show that lead has a very strong affinity to adsorb onto soil constituent surfaces such as clays, oxides, hydroxides, oxyhydroxides, and organic matter. Studies of ion exchange reactions of lead on a number of clay minerals such as montmorillonite, kaolinite, illite, and vermiculite showed that lead was preferentially adsorbed by exchange on clays, readily replacing calcium and potassium. Other studies on montmorillonitic, illitic, and kaolinitic soils confirmed that lead would preferentially exchange for calcium. Experimental data also showed that goethite (FeO•OH) and manganese-oxide minerals have a high adsorption affinity for lead.

The EPA (1999) review of lead K_d values reported in the literature for a number of soils led to the development of a look-up table for K_d values. The table is based on the observation that the two most important parameters affecting lead adsorption are pH and the equilibrium dissolved lead concentration. One of the three pH categories in the EPA look-up table is a range of 6.4 to 8.7, within which most of the FMSS groundwater pH measurements fall. Maximum lead concentrations in the overburden and bedrock samples are in the range of 16 to 28 µg/L, which fall within the range of 10 to 100 µg/L in the EPA look-up table. The range of lead K_d values provided in EPA (1999) under these site-specific pH and dissolved lead concentration ranges is 900 to 4,970 mL/g. A realistic K_d value to use for modeling purposes would be the geometric mean of these range limits, which is 2,115 mL/g.

2.1.5 Thallium

Like iron, thallium is also a redox-active element. However, it is dominantly present in the monovalent form in groundwater.. It will precipitate as a sulfide mineral under sulfate-reducing conditions. As long as the redox conditions are above the Eh-pH conditions where sulfide minerals are stable, the adsorptive behavior is most sensitive to pH, with K_d values increasing over a pH range from 5 to 8.

Estimates of thallium K_d values are provided in the *Soil Screening Guidance: Technical Background Document* (EPA,1996). Three K_d values of 44, 71, and 96 mL/g are provided for pH conditions of 4.9, 6.8, and 8.0, respectively. These experimentally determined K_d values were fit to a regression model as a function of pH, yielding the following relationship:

$$\mathrm{Log}_{10} K_d = 0.1093\ \mathrm{pH} + 1.1079$$

The pH of 38 groundwater samples from the bedrock aquifer ranges from 5.54 to 7.49, with a mean of 6.7. A set of 14 groundwater samples from the overburden aquifer has a very similar range, from 6.2 to 7.0, with a mean of 6.65. A realistic thallium K_d for the overburden and bedrock aquifer environments would thus be 69 mL/g, based on a mean pH of 6.7.

2.2 Organic Compounds

Adsorption coefficients for 2-chlorotoluene, methylene chloride, PCE, toluene, TCE, VC, and total xylenes are provided in **Table 3**. Adsorption coefficients for organic contaminants in groundwater are equal to the product of K_{oc}, and f_{oc}, the fraction of organic carbon that is present in the aquifer material.

$$K_d = f_{oc} \bullet K_{oc}$$

Values for K_{oc} are compound-specific, and are easily determined in the laboratory. The f_{oc} parameter is site-specific, and is calculated from the total organic carbon content of aquifer matrix material samples.

Experimentally determined K_{oc} values of many compounds have been published, and are often expressed in logarithmic units. One to three published K_{oc} values were found for the seven compounds of interest, as summarized in **Table 2**. If more than one K_{oc} was found, then the recommended value is based on the geometric mean of the published values. In the case of xylene, three published K_{oc} values were found for the *ortho*, *meta*, and *para* isomers. Since all three of the isomers are present in most fuel and solvent mixtures, the geometric mean of the three K_{oc} values is recommended.

The site-specific f_{oc} for the overburden is estimated at 1.6 percent (based on the average of five soil f_{oc} values). The published f_{oc} average for siltstone bedrock is 1.1 percent (Blatt, 1980). **Table 3** provides the recommended K_d values for overburden and bedrock aquifers based on these K_{oc} and f_{oc} values.

3.0 Methods for In-Situ Treatment of Contaminants in Site Groundwater

Several different remediation options have been considered and applied at sites where contamination of groundwater exists. Most of these remediation techniques involve manipulation of the redox state of the aquifer environment to cause the precipitation of metals, the adsorption of metals on mineral surfaces, or the enhancement of aerobic or anaerobic microbiological activity. The redox manipulation techniques are most effective for redox-active metals, which are metals that can exist in more than one valence state over the range of Eh, pH, temperature, and pressure conditions that exist in shallow groundwater environments. The different valence states of some of the metals of concern at the Site have very different solubilities and/or adsorption affinities, which translate into different mobilities. Other metals such as barium, beryllium, lead, and radium are not directly affected by redox conditions, but their mobilities can be indirectly affected. In some aquifer systems, the mobilities of beryllium and lead are controlled by sorption on iron and/or manganese oxides. These oxides are stable under oxidizing conditions, but if the redox potential falls below a threshold value (at sufficiently low pH), then these oxides can dissolve and can release any adsorbed trace elements to the groundwater. The mobilities of barium and radium are controlled in some aquifers by precipitation of barium sulfate (barite) and coprecipitation of radium sulfate, both of which have very low solubilities. Under sulfate-reducing conditions, sulfate gets reduced to sulfide, which drives the dissolution of these sulfate minerals and can release barium and radium to the groundwater.

Methods for increasing the redox potential of the aquifer include the injection of air, pure oxygen, ozone, hydrogen peroxide, and oxygen-releasing solid compounds such as magnesium peroxide and Oxygen Release Compound (ORC®, developed by Regenesis, 2004). More aggressive redox-increasing techniques include the injection of permanganate and persulfate compounds, and Fenton's reagent. These more aggressive techniques rapidly raise the redox potential to very high values in the treated areas.

Chemical techniques used to create reducing conditions include the injection of reducing agents such as calcium polysulfide, sodium thiosulfate, and proprietary compounds such as Metals Remediation Compound (MRC®, developed by Regenesis, 2004). Biological reduction techniques are most commonly used for in-situ treatment of organic contaminants, but have also been successfully used for metals. Biological reducing techniques involve the injection of compounds that provide a bio-available source of organic carbon thus stimulating anaerobic activity. Materials such as lactic acid, sodium lactate, molasses, cheese whey, emulsified soybean oil, and Hydrogen Release Compound (HRC® developed by Regenesis, 2004) have been successfully used for this purpose.

Caution should be exercised in the use of these redox manipulation techniques if more than one metal is of concern at a single location, because the mobilities of some metals have opposite behavior with respect to redox conditions. For instance, in high sulfate water, radium has a low mobility under oxidizing conditions, whereas uranium is the least mobile under reducing conditions. The in-situ application of a chemical oxidant in a reducing zone of an aquifer in which radium and uranium are present could have the effect of immobilizing radium, but mobilizing uranium. Adding a reducing agent to an oxic portion of an aquifer could have the opposite effect by mobilizing radium and immobilizing uranium.

3.1 Local Redox Conditions

Background groundwater conditions in the overburden layers at FMSS are oxidizing, and are oxidizing to mildly reducing in the bedrock units. However, within the benzene-impacted areas, redox conditions are strongly reducing, as evidenced by low to non-detectable dissolved oxygen; negative oxidation-reduction potential (ORP) measurements; conversion of nitrate to ammonia; and measurable dissolved iron, manganese, and methane. In addition, a large percentage of groundwater samples from monitoring wells located within the benzene plume had low but detectable concentrations of sulfide, indicating the presence of active sulfate-reducing anaerobes. Sulfate reduction only occurs under highly reducing conditions. High concentrations of dissolved sulfide are not expected in this environment because sulfide is quite reactive, and will combine with ferrous iron, as well as with trace elements such as arsenic, lead, and thallium, to precipitate as insoluble sulfide minerals. For this reason, the detection of even low levels of sulfide in groundwater usually indicates strongly reducing conditions.

3.2 Behavior and Treatment of Contaminants in Site Groundwater

Redox manipulation, via the creation of oxidizing or reducing conditions, is one of several remediation techniques that are under consideration at the Site. The introduction of oxygen to groundwater, via air sparging or the injection of ozone, hydrogen peroxide, or magnesium peroxide, has been shown to be effective in accelerating natural aerobic biodegradation rates of benzene, toluene, and xylene in systems where oxygen availability is the rate-limiting parameter. Oxygen addition may also be useful for causing in-situ adsorption of arsenic. The creation of reducing conditions may be useful in causing in-situ precipitation of arsenic, lead, thallium, and uranium. The effects of redox manipulation on all of the contaminants that are present within the plumes need to be considered. The effects of redox manipulation on the geochemical behavior of the metals arsenic, barium, beryllium, boron, chromium, iron, lead, lithium, radium, thallium, and uranium; and the organic compounds benzene, methylene chloride, PCE, toluene, TCE, VC, and xylene under various treatment scenarios, is discussed in the following paragraphs. A summary of the discussion with respect to metals is provided in **Table 4**.

Behavior and Treatment of Metals

Arsenic – Arsenic has two valence states (arsenite [+3] and arsenate [+5]) in the natural environment, so it qualifies as being directly redox-sensitive. Although there are two valence states, one really needs to think in terms of three redox zones when predicting arsenic mobility in groundwater, as summarized in **Table 5**. Under oxidizing conditions, inorganic arsenate is fairly immobile, because it speciates as oxyanions [$HAsO_4^{-2}$, $H_2AsO_4^-$] that strongly adsorb on the positively charged surfaces of iron oxides. Under intermediate redox conditions, arsenic will be present in the trivalent arsenite state, which speciates as a neutral form [$H_3AsO_3^o$ or $HAsO_2^o$]. Dissolved arsenite is soluble, and only weakly adsorbs to mineral surfaces because of its neutral charge. This form of arsenic is quite mobile. Under strongly reducing conditions where sulfate-reducing anaerobes are generating sulfide, arsenic will precipitate as one of several arsenic-sulfide minerals (orpiment, realgar, arsenopyrite) which have very low solubilities (Smedley and Kinniburgh, 2002; Sullivan and Aller, 1996). As a further complication, the adsorption of arsenate is pH dependent. Arsenate competes with hydroxy ions for sorption sites, resulting in increased mobility under pH conditions above approximately 8.5.

The addition of oxygen to the benzene-impacted areas may affect arsenic mobility in a few different ways, depending on the current conditions within the plume. If conditions within the benzene plume correspond to the "moderately reducing" redox conditions in **Table 5**, then arsenic is currently in the mobile trivalent arsenite form. The oxygen addition will oxidize dissolved arsenite to the less mobile pentavalent arsenate form. At the same time, dissolved ferrous iron, which is currently as high as a few tens of mg/L within the benzene plume, will rapidly oxidize to ferric iron and precipitate as $Fe(OH)_3$. Arsenate will adsorb on the $Fe(OH)_3$, and arsenic concentrations in groundwater will most likely return to background values. Alternatively, if current conditions within the benzene plume correspond to the "strongly reducing" conditions, and a significant mass of arsenic is currently present as sulfide minerals, then the oxygen addition will have a different effect. The addition of oxygen to a system containing metal-sulfide minerals results in the dissolution of the sulfide minerals and the generation of sulfuric acid. Arsenic released to groundwater by this process may remain mobile.

The amount (if any) of arsenic present as sulfide minerals within the benzene plume is not known, because these materials were not sampled and analyzed. Sulfide was detected in three of the nine groundwater samples analyzed for sulfide, including benzene plume Monitoring Wells BRPZ-9 (1.0 mg/L), BRPZ-4 (0.2 mg/L), and downgradient Monitoring Well MW-32D (0.5 mg/L). The presence of detectable sulfide in these groundwater samples suggest the presence of sulfide minerals within the benzene plume, but not necessarily arsenic-bearing sulfide minerals. It is recommended that the arsenic and sulfide content of solid material within the benzene plume be determined prior to the application of any oxidants to the plume.

Treatment of dissolved arsenite with an in-situ oxidant is predicted to be effective in causing oxidation to arsenate and subsequent adsorption on iron oxide surfaces. The oxidation rate of arsenite with a mild oxidant such as ORC® under these conditions is in the range of days to weeks, so a reasonably conservative treatment half-life is three days.

An alternative approach to in-situ treatment of arsenic is the use of MRC® or calcium polysulfide. The addition of calcium polysulfide provides a source of sulfide that will react with arsenic to precipitate an insoluble arsenic sulfide mineral. MRC® will hydrate and release an organosulfur compound that reacts to produce a stable metal-organosulfur complex. This complex sorbs strongly to soil and is immobile in the subsurface. Over time, the organic portion of the complex is biodegraded and the immobilized metals are incorporated into the soil matrix as sulfide solids. The sulfide solids tend to be stable under low redox potentials. The time required for arsenic to decrease to nondetectable concentrations in response to the injection of a reducing agent is in the range of days to weeks. A treatment half-life of 10 days in response to polysulfide or thiosulfate injection is a reasonable assumption.

Barium – Barium exists in groundwater as the divalent cation Ba^{+2}, and has similar geochemical properties as radium. Under oxidizing to mildly reducing conditions, its mobility is limited in many environments by the solubility of barite ($BaSO_4$), but those controls are removed under sulfate-reducing conditions when sulfate gets reduced to sulfide. Removal of barium in drinking water is usually performed in a treatment plant using lime softening, ion exchange, or reverse osmosis techniques. In-situ treatment of barium in groundwater is not a standard method, although permeable reactive barriers containing gypsum have been proposed at several sites.

Beryllium – Beryllium is always present in the divalent form in the natural environment, so it is not sensitive to redox conditions. Concentrations of beryllium in most aquifer systems are controlled by adsorption. EPA-approved treatment methods for the ex-situ removal of beryllium from water include activated alumina, coagulation/filtration, ion exchange, lime softening, and reverse osmosis. There is no proven in-situ method for immobilizing beryllium.

Boron – Boron is not redox-sensitive, and its mobility under the expected range of redox Eh conditions is high. Boron cannot be manipulated by changing redox conditions.

Chromium – Chromium can be present as the +3 or +6 valence states in the natural environment. The hexavalent form (chromate, or CrO_4^{-2}) is only stable under oxidizing conditions. This form is quite mobile due to high solubility and low affinity for adsorption. The trivalent form is immobile due to a combination of low solubility and high adsorption affinity (EPA, 1994). Creating a reducing environment is an effective way to immobilize hexavalent chromium. In-situ application of chemical or biological reducing agents has been shown to be effective in converting hexavalent chromium to the trivalent state and significantly lowering dissolved chromium concentrations.

Iron – Iron can exist in the divalent [ferrous, or Fe (II)] or trivalent [ferric, or Fe (III)] forms in aquifer environments. These two valence states have very different behavior. Dissolved iron concentrations in groundwater are controlled mostly by dissolution–precipitation reactions rather than adsorption–desorption reactions. The redox state of the aquifer environment is the most important parameter controlling iron concentrations in groundwater. Under oxidizing conditions where Fe (III) is stable, the solubility of iron is well below the standard analytical reporting limits. Detectable concentrations of iron in oxic groundwater samples are usually due to the presence of iron-bearing suspended particulates rather than dissolved iron. If the redox potential is below a critical value (which is a function of pH), then

Fe (II) becomes stable. The solubility of Fe (II), under reducing conditions, can be as high as several tens of mg/L.

The addition of organic contaminants such as benzene, fuel, or solvents to an aquifer initially stimulates aerobic activity, and results in the establishment of locally reducing conditions within the contaminant plume. Within the reducing zone, naturally occurring iron and manganese oxides that are part of the aquifer matrix become soluble and dissolve, leading to elevated iron and manganese concentrations. Trace metals such as arsenic, nickel, and vanadium, which naturally adsorb on iron oxides, can be released to the groundwater as the oxide minerals dissolve. Elevated iron, manganese, and trace element concentrations will remain elevated within this zone, as along as reducing redox conditions prevail.

A third condition corresponding to highly reducing redox potentials, where anaerobes are actively reducing sulfate to sulfide, needs to be considered as well. Under these conditions, iron reacts with sulfide produced by anaerobes to precipitate one or more sulfide minerals such as pyrite (FeS_2), pyrrhotite ($Fe_{1-x}S$), or marcasite (FeS_2) which have low solubilities.

The FMSS groundwater monitoring data illustrate the importance of redox in controlling iron concentrations. Iron concentrations are low outside of the benzene plume where redox conditions are mostly oxidizing, but are in the range of tens of mg/L within the plume. The zone within the plume is also characterized by low to nondetectable dissolved oxygen and strongly negative redox potentials.

It is important to take these processes into account when considering options for remediation of the organic contamination. Options that introduce oxygen into the aquifer to accelerate aerobic degradation of benzene (air sparging, ozone addition, ORC®, etc.), and the application of in-situ chemical oxidants (permanganate, persulfate, Fenton's reagent, etc.), will, as a side effect, rapidly oxidize and precipitate any excess iron and manganese that were mobilized within the reducing plume. Any trace elements that were also mobilized within the reducing zone will adsorb on the re-precipitated oxides, and usually return to background concentrations. Adding oxygen to a reducing zone where iron is soluble is a very effective method for precipitating iron. The only concern is that if iron concentrations are very high, and the permeability of the treatment zone is low, then plugging of the aquifer may result where iron precipitates.

Alternative remedial options that encourage reducing conditions, such as the application of lactic acid, molasses, soybean oil, or HRC®, will have the opposite effect on iron and manganese by promoting the dissolution of iron and manganese oxides in the aquifer matrix, thus raising dissolved concentrations, if moderately reducing conditions are established. If strongly reducing conditions are established, then iron may precipitate as sulfide minerals.

Lead – Dissolved lead is always divalent, so this metal is not directly affected by redox, but it can be indirectly controlled by redox under some conditions. Under oxic conditions, lead is soluble, but its concentrations in groundwater are usually limited by adsorption on the surfaces of clays, iron oxides, and manganese oxides (EPRI, 1984). Under sulfate-reducing conditions, lead will precipitate as a sulfide mineral (galena, PbS) that has a very low solubility. High concentrations of dissolved lead in an oxidizing environment can be lowered by injecting calcium polysulfide, or by creating sulfate-reducing anaerobic conditions. Lead concentrations will be controlled at low values under sulfate-reducing conditions by the solubility of lead sulfide. Low lead concentrations will persist, as long as the system remains reducing. A future return to natural oxic background conditions will result in the oxidation of sulfide minerals to yield sulfuric acid, accompanied by a release of lead into the solution.

Lead sulfide precipitation kinetics are fast (minutes to hours) in the laboratory. The rate-limiting step for in-situ lead immobilization in the field is subsurface mixing of the injected reagents with groundwater. Local site parameters such as porosity, permeability, and tortuosity (the average distance that water molecules travel between two points along a flow path relative to the linear distance between the points), and treatment parameters such as injection pattern and spacing, injection volumes, reagent concentration, and injection methods (surging, pulsing, recirculation, etc.), will all affect the actual treatment half-lives. Assuming that the treatment parameters are optimized for the site, a 10-day treatment half-life is a reasonable assumption for lead.

Lithium – This metal is not redox active, and will not be affected either directly or indirectly by redox manipulation. Lithium mobility is mostly controlled by adsorption on clay surfaces along the groundwater flow path. Lithium can be removed from water in a treatment plant via ion exchange or reverse osmosis, but no proven in-situ remedy is available.

Radium – Radium exists in groundwater as the divalent cation Ra^{+2}, and has similar geochemical properties as barium. Radium is not directly redox-sensitive, because it is always present in groundwater in the divalent state. However, radium mobility can be indirectly affected by redox. Radium concentrations in groundwater are often limited by the solubility of radium sulfate ($RaSO_4$), which is low. In systems where radium is controlled by $RaSO_4$, a solubility-product constraint exists where equilibrium radium concentrations are inversely proportional to sulfate concentrations. Under sulfate-reducing conditions, sulfate gets reduced to sulfide, so the solubility-limiting control of radium concentrations by $RaSO_4$ precipitation is no longer active, and radium can be mobile. Removal of radium in drinking water is usually performed in a treatment plant using lime softening, cation exchange, or reverse osmosis. In-situ treatment of radium in groundwater is not a standard method.

Thallium – Thallium is redox-sensitive and can exist in three different valence states (Tl^+, Tl^{+3}, and Tl^{+4}) under the range of environmental conditions. Under highly oxidizing conditions, it forms several different oxide minerals (Tl_2O_4, Tl_2O_3, Tl_2O) that limit thallium mobility. Under intermediate redox conditions, it is present in the monovalent state, and is fairly mobile. It will precipitate as a sulfide with a low solubility under sulfate-reducing conditions. Low concentrations will persist under sulfate-reducing conditions, as long as the system remains reducing. A future return to natural oxic background conditions will result in the oxidation of sulfide minerals to yield sulfuric acid, accompanied by a release of thallium into solution.

The injection of a reducing agent such as polysulfide, MRC®, or thiosulfate will rapidly precipitate thallium as a sulfide. The time required for thallium to decrease to nondetectable concentrations in response to the injection of a reducing agent is in the range of days to weeks. A treatment half-life of 10 days in response to polysulfide or thiosulfate injection is a reasonable assumption. Treatment half-lives for biological approaches, such as lactic acid or HRC® injection, are more uncertain because they are a function of several site-specific parameters, but a half-life of 40 days is a reasonable value for these techniques.

Uranium – Uranium has two valence states of +4 and +6, which have very different solubilities and mobilities. Uranium is fairly mobile under oxidizing conditions, where it exists as the hexavalent uranyl ion [UO_2^{+2}]. The uranyl ion forms a series of anionic carbonate complexes [$UO_2(CO_3)_2^{-2}$, $UO_2(CO_3)_3^{-4}$] that are soluble and do not strongly adsorb (Davis et al., 2004). Under reducing conditions, uranium will reduce to the tetravalent state, and will precipitate as the mineral uraninite (UO_2) which has very low solubility. The differences in uranium solubility under oxidizing versus reducing conditions can be as great as a factor of 1×10^5.

Insight into uranium behavior can be gained by considering the natural occurrences of the element. Uranium deposits in Colorado and Wyoming formed when oxic surface water carrying small amounts of dissolved uranium recharged aquifers that had local reducing conditions. Uranium precipitated to form commercial deposits at locations along the aquifer flow path, where the redox conditions fell below a threshold value below which uranium reduced from the hexavalent to the tetravalent state.

Some of these deposits have been exploited using in-situ mining techniques that reverse the natural redox conditions. A dilute solution of hydrogen peroxide and sodium carbonate is pumped into the ore zone. The hydrogen peroxide oxidizes the uranium to the hexavalent state, and the carbonate acts as a complexing agent that forms soluble uranyl-carbonate species. The injected solution is then pumped back to the surface where the uranium is extracted by anion exchange, and the reagents are recycled.

Imposing reducing conditions in a uranium-contaminated aquifer can be very effective in immobilizing uranium, as long as the redox remains low. The application of chemical or biological reducing agents to cause the precipitation of uranium under reducing conditions should be effective, although this is not a proven technology. Injection of chemical reducing agents, such as polysulfide or thiosulfate, should cause the rapid precipitation of uranium, with treatment half-lives in the range of 10 days. Treatment half-lives for biological approaches, such as lactic acid or HRC® injection, are more uncertain because they are a function of several site-specific parameters, but a half-life of 40 days is a reasonable value for these techniques. The injection of MRC® may be effective for immobilization of uranium also, although there is no data available on it's effectiveness for this application. Uranium does not precipitate as a sulfide, but MRC® promotes a highly reducing environment that will cause the precipitation of uranium as an insoluble oxide. The time required for uranium to decrease to nondetectable concentrations in response to the injection of a reducing agent would probably be in the range of days to weeks. A treatment half-life of 10 days in response to polysulfide or thiosulfate injection is a reasonable assumption.

Behavior and Treatment of Organic Compounds

The seven organic constituents (benzene, methylene chloride, PCE, toluene, TCE, VC, and xylene) are discussed in three groups below.

Benzene, Toluene, and Xylene – Benzene is a key constituent at FMSS. Toluene and xylene are also present at lower concentrations, and have chemical properties and degradation rates that are similar to benzene. All three compounds are in the "aromatic hydrocarbon" family, and consist of a benzene ring with varying degrees of methylation (toluene has one methyl group and xylene has two methyl groups). These three compounds, along with ethylbenzene, are major components of automotive fuel, and have been extensively studied at sites where fuel releases have occurred.

The microbial degradation of benzene, toluene, and xylene occur by the same pathways which are classified as aerobic or anaerobic. The three compounds will degrade fastest under aerobic conditions, but oxygen is consumed and rapidly depleted during this process in the absence of any active measures to resupply oxygen to the system (air sparging, ozone or hydrogen peroxide addition, ORC® injection, etc.). Once oxygen is depleted by the aerobes, then anaerobic microbes will take over, and degradation will continue at some rate by nitrate-reducing, manganese-reducing, iron-reducing, sulfate-reducing, or methanogenic anaerobes. These degradation processes generally occur in sequence, and one process yields to the next as the required nutrients are depleted. Following a benzene release, aerobic degradation will initially be dominant until the dissolved oxygen is consumed and falls below a concentration of 0.5 mg/L. Nitrate-reducing anaerobes then dominate, and will be effective at some rate until the available nitrate gets consumed. At that point, manganese- and/or iron-reducing anaerobes will dominate until the ferric iron and oxidized forms of manganese are converted to ferrous iron and reduced forms of manganese. At that point, sulfate-reducing anaerobes will reduce sulfate to sulfide until most of the dissolved sulfate is reduced. Methanogenic anaerobes will then dominate the degradation process, and will proceed at some rate until the available organic carbon is converted to methane.

Each of these steps is characterized by progressively lower redox potentials, which is measured as ORP in the field. Each step is also characterized by the sequential depletion of dissolved oxygen, conversion of nitrate to nitrogen or ammonia, increases in dissolved manganese and iron, conversion of sulfate to sulfide, and finally, conversion of carbon dioxide and organic carbon to methane.

The GWRI Report provides data on the ORP and concentrations of these microbial indicators, including dissolved oxygen, nitrate, ammonia, dissolved manganese, dissolved iron, sulfate, sulfide, and methane in samples from benzene-impacted and non-impacted locations. Results from the benzene-impacted locations clearly show lower ORP, corroborated by oxygen depletion, conversion of nitrate to ammonia, solubilization of manganese and iron, and methane production.

These results indicate that anaerobic conditions are firmly established within the benzene-impacted area, so the local benzene degradation rates will be limited by the rates that are specific to these various anaerobe populations. Remedial methods that supply the system with oxygen by air sparging, ozone injection, hydrogen peroxide injection, and the use of oxygen-releasing compounds such as magnesium peroxide and ORC®, are standard techniques implemented at sites impacted by benzene, toluene, and xylene. These techniques return the system to oxic conditions under which the faster aerobic degradation processes become re-established. Many sites where these techniques have been implemented have experienced greatly increased benzene, toluene, and xylene degradation rates.

Degradation rates are highly site-specific, but do provide an estimate of what can be expected after these remediation techniques are applied. The actual rate increases that will be observed at a given site cannot be accurately predicted, but positive improvements are expected. Published case histories from these sites, where enhanced biodegradation by oxygen addition was employed, can serve as a qualitative guide (Bohan and Schlett, 1997; Chapman et al., 1997; Duffy et al., 1999; Fischer et al., 2001; Hicks et al., 2001; Johnson, et al., 1997; Kao and Borden, 1994; Muniz et al., 2001; Mysona and Hughes, 1999; Oudijk, et al., 1999; Reed et al., 2001).

Published case histories regarding the effects of oxygen addition techniques on the degradation rate of benzene were evaluated for applicability to the FMSS. Benzene was conservatively selected to represent the three aromatic hydrocarbons, because benzene contamination is more widespread than toluene or xylene at the FMSS, and the degradation rate of benzene is lower than toluene and xylene (Lu and Zheng, 2003).

Most of these case histories express degradation in terms of a percentage or concentration decrease after a specific period of time. Contaminant fate and transport models require degradation rates, usually in the form of half-lives, that are used to predict the effects of enhanced degradation on the size and shape of the groundwater plume as a function of time. If the concentrations at three or more points in time are provided in a case history, then a degradation rate expressed as a half-life can be calculated from the data, based on an exponential decay model.

Several investigations have demonstrated that the biodegradation of fuel hydrocarbons can be approximated by first-order rate constants that can be expressed as half-lives (Wiedemeier et al., 1995 and 1996). To convert benzene concentration versus time data to a degradation half-life, benzene concentrations were first plotted on a time-trend graph. The trends were then fit with an exponential least squares decay curve of the form:

$$y = N_0 e^{-kt}$$

where: y = concentration

N_0 = concentration at $t = 0$

e = base of the natural logarithm

k = decay rate

t = time

The decay rate is equal to:

$$k = \frac{1}{t} \ln \frac{N_0}{N}$$

where N is the concentration after time t. The half-life ($t_{1/2}$) of the decay process is the time after which the original concentration has decayed to one-half of it's value, or $N = N_0/2$. The half-life is thus related to the decay rate as:

$$t_{1/2} = \frac{\ln 2}{k} = \frac{0.693}{k}$$

The correlation coefficient (R^2) provided with each of the calculated half-lives is a measure of the goodness-of-fit, or the degree to which the data deviate from the fitted exponential decay curve. The R^2 parameter ranges from 0.0 to 1.0. An R^2 of 1.0 indicates a perfect fit in which the curve passes through each data point, and an R^2 of 0.0 indicates no correlation between the data and an exponential decay curve.

Example: One applicable case history is the use of ORC® at a former gasoline service station that had gasoline subsurface free product contamination (Mysona and Hughes, 1999). The ORC® compound is a proprietary form of slow-releasing magnesium peroxide (MgO_2), which releases oxygen by the reaction:

$$MgO_2 + H_2O \rightarrow \tfrac{1}{2}O_2 + Mg(OH)_2$$

Results of five groundwater sampling events conducted at three monitoring wells during the remediation operations at the service station provide a good data set for the calculation of enhanced benzene degradation rates. **Table 6** provides the calculated treatment half-lives and correlation coefficients of the exponential fit to the benzene monitoring data at three wells. The correlation coefficients of the exponential fit indicate good agreement with an exponential decay model. The half-lives at the three wells, calculated from five samples from each well obtained over a 1.3 year monitoring period, range from 0.29 to 0.76 years, with a mean of 0.51 years. In conclusion, a benzene half-live degradation rate in the range of six months has been observed at three locations where oxygen addition has been used to enhance the natural degradation rates.

Methylene Chloride – Methylene chloride, also known as dichloromethane, is a colorless liquid that has a mild sweet odor, evaporates very quickly, and will not easily burn. It is widely used as an industrial solvent and as a paint stripper. It can also be found in certain aerosol and pesticide products, and is used in the manufacture of photographic film. Methylene chloride does not appear to occur naturally in the environment. It is also an intermediate anaerobic degradation product of carbon tetrachloride that eventually further degrades to methane (Davis, et al. 2003).

Rapid biodegradation rates have been observed under laboratory-controlled aerobic and anaerobic conditions, but actual rates vary under natural conditions (ATSDR, 2004). Regenesis claims that HRC® is effective in promoting the dechlorination of methylene chloride to methane under anaerobic conditions, and can be treated in a similar manner as PCE, TCE, and VC.

PCE, TCE, and VC – The compounds PCE, TCE, dichloroethene (DCE), and VC are part of a degradation series. Under favorably reducing conditions, anaerobic microbial activity will sequentially replace a chlorine atom with a hydrogen atom to yield the reductive dechlorination sequence:

PCE → TCE → *cis*-1,2-DCE → VC → Ethene

Ethene will eventually degrade to carbon dioxide and water. The compounds PCE and TCE are commonly used industrial cleaners and solvents, but *cis*-1,2-DCE and VC are mostly used in the manufacturing of plastics, and are not usually used at other industrial facilities. Therefore, the presence of *cis*-1,2-DCE and VC indicate that degradation is occurring via reductive dechlorination (EPA, 1998). Eventually, all of the compounds will degrade to nondetectable concentrations when these sequential reactions go to completion; but temporary increases in daughter products are expected as the parent and daughter compounds degrade at different rates. In fact, temporary increases in daughter products provide evidence that microbial degradation is effectively proceeding.

Each of the sequential steps requires progressively lower redox conditions. If only moderately reducing conditions are achieved, then the reductive dechlorination reaction sequence may stall, resulting in conversion of the original solvents to DCE or VC with no further degradation. A common limiting factor in the establishment of highly reducing conditions at many sites is the concentration of bio-available forms of organic carbon that anaerobes can utilize. EPA (1998) guidance suggests that total organic carbon concentrations of at least 20 mg/L are required to achieve the reducing conditions necessary for complete degradation of the sequence of compounds via anaerobic reductive dechlorination pathways. At sites that are limited by natural organic carbon concentrations, the injection of some bio-available form of carbon, such as molasses, unrefined soybean oil, sodium lactate, lactic acid, or HRC®, can be effective in driving the redox potential to lower values, thus promoting the complete degradation of the compounds.

It should be noted that VC is the only compound in the sequence that can also be degraded at a reasonable rate under aerobic, as well as anaerobic, conditions. Taking advantage of this can be problematic, because it may not be possible to maintain the reducing conditions necessary for degradation of all of the PCE, TCE, and DCE to VC, while maintaining oxic conditions for degradation of VC. This has been achieved at some sites in a two phase process, where reducing conditions are maintained for a period of time to allow conversion to VC, followed by the imposition of oxic conditions in a later phase to degrade the VC.

3.3 Summary and Recommendations for In-Situ Treatment of Contaminants in Site Groundwater

In-situ remediation of benzene, toluene, and xylene can be accomplished by oxygen-addition techniques that stimulate aerobic microbial activity such as the application of ozone, peroxide, or ORC®. Remediation of the chlorinated compounds PCE and TCE is best accomplished by stimulating anaerobic microbial activity thorough the application of bioavailable sources of organic carbon (lactic acid, soybean oil, etc), or proprietary compounds such as HRC®. VC can be remediated by encouraging either aerobic or anaerobic conditions. However, anaerobic techniques are usually more appropriate for VC because it is a degradation product of PCE and TCE, so it is usually co-located with these compounds.

A summary of the previous discussions for metals is provided in **Table 4**. The terms *high* and *low* used in the table to describe the mobility of the metals are relative terms. A high potential for mobility does not imply that high concentrations will actually be present. It is certainly possible to have high mobility and low concentrations, if the local concentrations are limited by availability of the metal.

It is apparent from **Table 4** that there is no unique set of conditions that will limit the mobilities of all of the metals of concern, because some of the metals have opposite behavior with respect to their mobilities under different redox conditions. The following recommendations follow from this discussion.

1. In the selection of remedial options for metals in groundwater, it is recommended that the Site be considered as consisting of two zones: an oxic zone that is outside of the benzene plume, and a reducing zone that is within the benzene plume. Within each of the zones, the metals of key concern should be prioritized because it may not be technologically feasible to treat all of the metals, especially those with opposite redox behavior.

2. The addition of oxygen is being considered to accelerate the natural attenuation of benzene within the benzene plume. It is recommended that the arsenic, lead, thallium, uranium, and sulfide content of solid material within the benzene plume be determined prior to the application of oxygen to the plume. If a significant mass of these metals has precipitated within the benzene plume as sulfide minerals, then the addition of oxygen to accelerate the natural attenuation of benzene may have the unwanted side effect of mobilizing these metals.

3. An additional point to consider is that any in-situ remedy expected to be effective on a long-term basis should be compatible with natural background conditions that will exist over the long-term. For instance, consider the case of injecting calcium polysulfide or creating in-situ sulfate-reducing conditions by stimulating anaerobes within an otherwise oxic aquifer. The solubilities of several metals, including arsenic, lead, and thallium, will be maintained at very low concentrations as long as the Eh-pH conditions remain below the sulfate/sulfide boundary. However, if the redox buffer capacity of the system becomes exhausted and oxic conditions return at some point in the future, then the sulfide minerals will oxidize to yield sulfuric acid, and the sequestered metals will be remobilized. Likewise, if in-situ oxidation is applied at a site where the natural background redox conditions are low [below the Fe(III)/Fe(II) curve in Eh-pH space], then manganese oxides and iron oxides could dissolve when low redox conditions return, and any adsorbed trace metals could then be released back to solution. On a long enough time scale, the system will eventually return to its original redox state. If the natural conditions are oxidizing, then in-situ sulfide precipitation is only a temporary remedy. Likewise, if the natural conditions are reducing, then in-situ oxidation is only a temporary remedy, although these temporary conditions may persist for a significant period of time.

4.0 Radionuclide Half-Lives and Specific Activities

Each radionuclide has a unique half-life, which is the time required for a given mass of the radionuclide to be reduced by one-half as a result of radioactive decay. Each radionuclide also has a unique specific activity, which is defined as the radioactivity (number of disintegrations per unit time) released by a mass of one gram. The Curie is a unit of radioactivity originally based on the disintegration rate of one gram of Ra-226, but is now defined as the quantity of any radioactive nuclide in which the number of disintegrations per second is 3.7×10^{10} (Friedlander et al., 1981). Half-life and specific activity are related by the relationship:

$$SA \times t_{1/2} \times M = 1.1283 \times 10^{13}$$

where:

SA = Specific activity (Ci/g)

$t_{1/2}$ = Half-life (seconds)

M = Atomic mass of the nuclide (g/mole)

Half-lives and specific activities for Ra-226, Ra-228, Th-228, Th-230, Th-232, U-234, U-235, and U-238 are provided in **Table** 7. Specific activities are useful for converting activities to mass in transport calculations.

5.0 Evaluation of the Sources of Gross Alpha and Gross Beta Activity in Groundwater

This section provides the methodology and results of a statistical evaluation of the gross alpha, gross beta, and other radiological analyses of GWRI Phase II bedrock groundwater samples. The objective of the evaluation is to identify the principal sources of gross alpha and gross beta activities in groundwater samples.

The Federal/New Jersey Maximum Contaminant Level (MCL) for gross alpha is 15 pCi/L, which excludes uranium and radon activities. Radon was not analyzed in groundwater samples, so its contributions to the gross alpha measurements is unknown. The Federal/NJDEP gross beta screening value is 50 pCi/L, which excludes the contributions from naturally occurring K-40.

One of the overburden groundwater samples exceeded the gross alpha MCL, and none of them exceeded the gross beta MCL. The only gross alpha exceedance in the overburden samples (292.91 J [±131.71] pCi/L) was detected at Monitoring Well OBMW10. A large measurement error was associated with this sample, due to the small sample volume analyzed and high total dissolved solids present in the sample. A larger number of bedrock samples exceeded both the gross alpha and gross beta MCLs. Three gross alpha exceedances were detected in bedrock groundwater samples. The maximum detected gross alpha result was present in the groundwater sample obtained from Monitoring Well BRPZ-5 (57.47 pCi/L), which is located in Former Retention Pond C on the MISS. Exceedances were also detected at Monitoring Wells BRPZ-2 (27 pCi/L) and MW-3D (18.42 pCi/L). Four samples contained adjusted gross beta at a concentration exceeding the screening standard, including MISS Monitoring Wells BRPZ-2RE (168.02 pCi/L), BRPZ-5 (53.41 pCi/L), MW-26D (117.51 pCi/L), and off-site Monitoring Well MW-13D (68.98 pCi/L).

It has been suggested that gross alpha and gross beta be treated as contaminants and simulated using a fate and transport model, along with other contaminants. However, if it can be shown that there are several significant contributors to these gross measurements (i.e. if isotopes of uranium, radium, thorium, and other radioelements all contribute more or less equally), then gross alpha or gross beta cannot be modeled as a contaminant, because these elements have very different transport properties. Alternatively, if it can be shown that a significant fraction of the gross measurements can be attributable to a single radioelement, then the gross measurements do not need to be modeled, because the key radioelement that is contributing to the gross measurements has already been modeled.

Gross alpha and gross beta measurements are expected to exceed the sum of the analyzed alpha (α) and beta (β) emitters, because of the presence of many short-lived daughters that were not analyzed. For instance, in the U-238 decay chain:

> U-238(α) → **Th-234(β)** → **Pa-234(α)** → U-234(α) → Th-230(α) → Ra-226(α) → **Rn-222(α)** → **Po-218(β)** → **Pb-214(β)** → **Bi-214(β)** → **Po-214(α)** → **Pb-210(β)** → **Bi-210(β)** → **Po-210(α)** → **Pb-206(stable)**

for the Th-232 chain:

> Th-232(α) → Ra-228(β) → **Ac-228(β)** → Th-228(α) → **Ra-224(α)** → **Rn-220(α)** → **Po-216(α)** → **Pb212(β)** → **Bi-212(α)** → **Po-212(α)** → **Pb-208(stable)**

and for the U-235 chain:

> U-235(α) → **Th-231(β)** → **Pa-231(α)** → **Ac-227(β)** → **Th-227(β)** → **Ra-223(α)** → **Rn-219(α)** → **Po-215(α)** → **Pb-211(β)** → **Bi-211(α)** → **Tl-207(β)** → **Pb-207(stable).**

The eight radionuclides that were analyzed are shown in bold. Gross measurements, even after subtracting contributions from U-234(α), U-235(α), U-238(α), and K-40(β), will still get contributions from many of the other α and β emitters in these decay chains, if they are present at secular equilibrium (a condition where the daughters of a long-lived parent achieve activities that are equal to the activity of the parent). The extent of these contributions depends on the degree of secular equilibrium that has been established between the daughters, and the extent to which the aquifer acts as a closed system (i.e. does radon [Rn] escape as a gas).

Although the sum of the analyzed emitters (bold above) will be less than the gross measurements, the activities of a parent should be correlated with the gross measurements, if that parent (and associated daughters) was the dominant contributor to the gross measurements.

5.1 Methodology

Correlations between the gross measurements and the individual analyzed radionuclides were evaluated to determine if most of the gross activity can be attributed to a main contributor, or if multiple radioelements are all significant contributors. Analyses of the Phase II GWRI samples from the bedrock aquifer were used in the evaluation. Analyses of samples from the overburden aquifer were not used, because activities were lower, percentages of nondetects were higher, there were no exceedances of the gross beta MCL, and only one exceedance of the gross alpha MCL in the overburden data set.

Correlation is a measure of the relationship between two or more variables. The *correlation coefficient* is a measure of the extent to which values of two variables are proportional or linearly related to each other. The most widely-used type of correlation coefficient is Pearson *R*, also called linear, or product-moment correlation. The Pearson product moment correlation coefficient is calculated as:

$$R_{12} = \Sigma(Y_{i1} - \mu_1) * (Y_{i2} - \mu_2) / [\Sigma(Y_{i1} - \mu_1)^2 * \Sigma(Y_{i2} - \mu_2)^2]^{(1/2)}$$

where:

R_{12} = Correlation coefficient between variables 1 and 2

Y_{i1} = The i^{th} value of variable 1

Y_{i2} = The i^{th} value of variable 2

μ_1 = Mean of variable 1

μ_2 = Mean of variable 2

Correlation coefficients for a pair of variables can range from -1.00 to +1.00. The value of -1.00 represents a perfect negative correlation, while a value of +1.00 represents a perfect positive correlation. A value of 0.00 represents a lack of correlation.

5.2 Results

A table of correlation coefficients between gross alpha and gross beta versus the activities of the specific radionuclides that were analyzed in the Phase II GWRI bedrock groundwater samples is provided in **Table 8**. Summary statistics, including the number of samples, percent non-detects, minimum, median, mean, and maximum activities for the radiological parameters are provided in **Table 9**. Results for gross beta and gross alpha are discussed below.

Gross Beta – Elemental potassium and K-40 are the only parameters that are highly correlated with gross beta. The K-40 values are calculated by multiplying the elemental potassium concentration by a constant representing the fraction of total potassium present as K-40. Elemental potassium and K-40 are thus perfectly correlated with each other, so both parameters show the same degree of correlation with gross beta. The correlation coefficient of 0.99 between gross beta and K-40 suggests that the variance in gross beta is almost entirely due to the variance in potassium concentrations. Calculated K-40 activities in the bedrock groundwater samples have a considerable range of 0.62 (±0.12) to 593.49 (±119) pCi/L, or nearly three orders of magnitude, with a mean of 45.46 (±9.1). Measured gross beta activities in the bedrock samples range from 0.02 (±1.35) to 711 (±0.84), with a mean of 52.96 (±10.3) pCi/L, which is a similar range.

The relationship between calculated K-40 and gross beta activities is shown on **Figure 1**. The high degree of correlation is evident in the plot. The same data is shown on **Figure 2**, along with error bars indicating the uncertainties of the point positions, and a diagonal line with a slope of unity. If all of the gross beta was due to K-40, then one would expect a slope of unity. This perspective shows that the line with a slope of unity passes through the points, when the uncertainties are considered.

The only other beta emitter besides K-40 analyzed was Ra-228. Radium-228 activities in the 54 bedrock samples range from 0.05 to 2.08 pCi/L, and 40 of those measurements were considered to be nondetects. Activities attributable to Ra-228 are less than the uncertainties of the K-40 and gross beta measurements, indicating that Ra-228 is not a significant contributor to gross beta activities. In conclusion, activities detected by the gross beta measurements are mostly due to the presence of elemental potassium in the groundwater.

Adjusted Gross Beta – Adjusted gross beta is defined as a gross beta measurement minus the contribution from K-40. There were four exceedances of the adjusted gross beta criteria of 50 pCi/L at the following bedrock aquifer Monitoring Wells: MW-13D (68.98 pCi/L), BRPZ-2RE (168.02 pCi/L), BRPZ-5 (45'-55') (53.41 pCi/L), and MW-26D (117.51 pCi/L). One explanation for these exceedances is that other beta emitters besides K-40 are present in the samples. However, the uncertainties of these adjusted activities are fairly high. **Table 10** provides the gross beta, elemental potassium, calculated K-40 activity, and adjusted gross beta results for the four samples in question. Uncertainties in these parameters are also provided in the table. Measurement errors for gross beta are provided by the laboratory, and range from ± 4 percent to ± 24 percent of the reported activities in the four samples. The post-analysis data validation process assigned the analysis of the sample from Monitoring Well BRPZ-2RE, which had the highest gross beta activity of 473.16 (±39.98) pCi/L, a J-flag, indicating that the results are estimated, and are more uncertain than the laboratory-provided uncertainty.

The K-40 activities are calculated based on the elemental potassium in the samples. Uncertainties in the potassium concentrations were not provided by the laboratory, but are generally considered to be in the range of ±15 to 20 percent. Two of the four potassium concentrations are J-flagged, indicating that the uncertainties in these results may be higher than 20 percent. The uncertainty (σ) in a value c that is calculated by the difference between two uncertain values a and b is equal to:

$$\sigma_c = (\sigma_a^2 + \sigma_b^2)^{1/2}$$

Adjusted gross beta activities and their uncertainties calculated by this method are provided in **Table 10** for the four samples in question. The samples from Monitoring Wells BRPZ-5 and MW-26D actually have greater uncertainties than the reported activities themselves. The sample from Monitoring Well BRPZ-2RE has the highest adjusted gross beta activity of 168.02 (±72.96) pCi/L; however, both the gross beta and potassium results are qualified with J-flags, so this calculated activity is highly uncertain. The remaining sample from Monitoring Well MW-13D has an adjusted gross beta activity of 68.98 (±4.76) pCi/L. Other beta emitters that were not analyzed for may be present in this sample, although none of the analyzed radiological parameters were elevated.

Gross Alpha – Gross alpha (GA) measurements range from 0.0 (±2.98) to 67.74 (±20.25), with a mean of 8.58 (±11.1) pCi/L. The measurements are correlated most strongly with U-234, U-238, and total uranium, as shown on **Table 8**. (Total uranium is the calculated sum of U-234, U-235, and U-238 activities.) The correlation of total uranium versus GA is shown on **Figure 3**. Activities of U-234 and U-238, which are members of the same decay chain, are the dominant contributors to GA activities. Uranium-235 activities are mostly nondetectable (91 percent not detected). The maximum reported U-235 activity of 0.7 pCi/L was considered to be a nondetected result.

The three thorium isotopes (Th-228, Th-230, and Th-232) are all alpha emitters, but they show poor correlations with GA (R^2= 0.14, 0.17, and 0.04, respectively). In addition, the range of activities of the three thorium isotopes in the bedrock groundwater samples are much lower than the GA activities, as shown in **Table 9**. Radium-226 is an alpha emitter that was analyzed, but the low GA correlation coefficient of 0.16, and the maximum Ra-226 activity of only 2.28 pCi/L, indicate that it is not a significant contributor to GA.

Adjusted Gross Alpha – Adjusted gross alpha (AGA) activity is defined as the GA activity minus any contributions from isotopes of uranium and radon. The Federal/New Jersery MCL for AGA is 15 pCi/L. Radon was not analyzed, so its effect on the GA value is unknown. As a result, the AGA activities may be biased high, because of the unknown contributions from Rn-220 and Rn-222, which are daughters of the U-238 and Th-232 decay chains, respectively. AGA is defined in the absence of radon data as GA minus the uranium contributions, or:

$$\text{AGA} = \text{GA} - {}^{234}\text{U} - {}^{235}\text{U} - {}^{238}\text{U}$$

The uncertainty in the calculated AGA is equal to

$$\sigma_{AGA} = (\sigma_{GA}^2 + \sigma_{U\text{-}234}^2 + \sigma_{U\text{-}235}^2 + \sigma_{U\text{-}238}^2)^{1/2}$$

An additional, but unquantifiable, component of error is from R-220 and Rn-222 values which were not determined or subtracted from GA. Three AGA exceedances of the MCL were detected in bedrock groundwater samples. Gross and AGA activities, uranium isotope activities, and associated uncertainties are provided for these three samples in **Table 11**. The maximum AGA result of 57.47 (±20.37) pCi/L was present in the groundwater sample obtained from Monitoring Well BRPZ-5, which is located in Former Retention Pond C on the MISS. A second exceedance of 27.0 (±25.1) pCi/L was detected at well BRPZ-2. This result is highly uncertain when the measurement errors are considered. The third exceedance at Monitoring Well MW-3D of 18.42 (±17.2) pCi/L is also highly uncertain when measurement errors are considered. In fact, the GA measurement, and all three uranium isotope measurements in the Monitoring Well MW-3D sample, are considered to be nondetected.

The only statistically significant AGA exceedance of the MCL is the sample from Monitoring Well BRPZ-5 of 57.47 (±20.37) pCi/L, which is located in Former Retention Pond C on the MISS. Other alpha emitters that were not analyzed may be present in this sample, although none of the analyzed alpha emitters (Ra-226 and three thorium isotopes) were elevated.

5.3 Conclusions

Evaluations of the correlations of analyzed radionuclides versus GA and gross beta activities, and comparisons of the ranges of activities for all of the radiological parameters were performed. Results indicate that K-40 is the dominant contributor to gross beta activities; and that U-234 and U-238, which are members of the same decay chain, are the dominant contributors to GA activities. Based on the observed correlations, GA will closely follow the movement of uranium in the bedrock aquifer. The only statistically significant exceedance of AGA was observed at Monitoring Well BRPZ-5, which is in the vicinity of the Former Retention Pond source area. This indicates that one or more non-uranium alpha emitters are present in the bedrock groundwater only at this source area location. This observation suggests that the mobility of the alpha emitters are low.

Gross beta will closely follow the movement of elemental potassium in the aquifer. The only statistically significant exceedance of adjusted gross beta was at Monitoring Well MW-13D, which is due to the presence of one or more beta emitters other than K-40.

6.0 References

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TABLE 1

Adsorption Coefficients for Barium, Beryllium, Lead, and Thallium in Overburden and Bedrock Aquifers

Element	Adsorption Coefficients[1]	Substrate	Source	Range[2]
Barium	K_d= 66	Unweathered Glacial Till	IT, 1993	K_d= 41 to 128 **Recommended value = 70**
	K_d= 128	Weathered Glacial Till	IT, 1993	
	K_d= 41	pH = 6.8	EPA, 1996	
Beryllium	K_d = 790	pH = 6.8	EPA, 1996	K_d= 790 to 800 **Recommended value = 790**
	K_d = 800	Loam soil	Sheppard and Thibault,1990	
Lead	K_d= 19	Sand Soil, minimum	Sheppard and Thibault,1990	K_d= 19 to 4970 **Recommended value = 2115**
	K_d= 270	Sand Soil, geometric mean (n=3)	Sheppard and Thibault,1990	
	K_d= 1405	Sand Soil, maximum	Sheppard and Thibault,1990	
	K_d= 900	Minimum for site pH and Pb G.W. Conc.	EPA, 1999	
	K_d=4970	Maximum for site pH and Pb G.W. Conc.	EPA, 1999	
	K_d = 2115	Site-specific pH and Pb concentration	EPA, 1999	
Thallium	K_d= 44	pH = 4.9	EPA, 1996	K_d= 44 to 96 **Recommended value = 69**
	K_d= 71	pH = 6.8	EPA, 1996	
	K_d= 96	pH = 8.0	EPA, 1996	

(1) Adsorption coefficients in units of mL/g.

(2) The recommended value for use in the contaminant transport model is based on the geometric mean of the published values.

TABLE 2

Organic Carbon Adsorption Coefficients for Seven Organic Compounds

Compound		Log K_{oc} (mL/g)	K_{oc} (mL/g)	Recommended K_{oc}* (mL/g)	Reference
2-Chlorotoluene		2.28	191	191	Chiou et al., 1979
Methylene Chloride		0.94	8.7	8.7	Schwille, 1988
Tetrachloroethene		2.42	263	272	Abdul et al., 1987
		2.56	363		Schwille, 1988
		2.322	210		Chiou et al., 1979
Toluene		2.06	115	132	Abdul et al., 1987
		2.18	151		Garbarini and Lion, 1986
Trichloroethene		1.81	64.6	95	Abdul et al., 1987
		2.1	126		Schwille, 1988
		2.025	106		Garbarini and Lion, 1986
Vinyl Chloride		0.39	2.5	2.5	Karickhoff, 1979
Xylene	ortho	2.11	129	347	Abdul et al., 1987
	meta	3.2	1585		Abdul et al., 1987
	para	2.31	204		Abdul et al., 1987

* Based on geometric mean of referenced values

TABLE 3

Adsorption Coefficients for Seven Organic Compounds

Compound	**Recommended** K_{oc} **(mL/g)**	**Overburden** Kd* **(mL/g)**	**Bedrock** Kd** **(mL/g)**
2-Chlorotoluene	191	3.05	2.10
Methylene Chloride	8.7	0.14	0.10
Tetrachloroethene	272	4.35	2.99
Toluene	132	2.11	1.45
Trichloroethene	95	1.52	1.05
Vinyl Chloride	2.5	0.039	0.027
Total Xylenes	347	5.55	3.81

* Based on an f_{oc} of 1.6%

** Based on an f_{oc} of 1.1%

TABLE 4

Summary of the Mobilities of Elements Under Varying Redox Conditions

Element	Oxidizing	Intermediate	Sulfate-Reducing
Arsenic	Low	High	Low
Barium	Low	Low	High
Boron	High	High	High
Chromium	High	Low	Low
Iron	Low	High	Low
Lead	High	High	Low
Lithium	High	High	High
Radium	Low	Low	High
Thallium	Low	High	Low
Uranium	High	High	Low

TABLE 5

Mobility of Arsenic Under Different Redox Conditions

Redox Conditions	Arsenic Mobility	Arsenic Valence	Arsenic Species	Sulfur Species	Arsenic Precipitates
Oxidizing	Low	+5	$HAsO_4^{-2}$, $H_2AsO_4^-$	SO_4^{-2}	None
Moderately Reducing	High	+3	$HAsO_2^o$, $H_3AsO_3^o$	SO_4^{-2}	None
Strongly Reducing	Low	+3	$HAsO_2^o$, $H_3AsO_3^o$	H_2S, HS^-	As-Sulfides

TABLE 6

Enhanced Benzene Degradation Rates at a Gasoline Service Station Site

(Calculated from the data of Mysona and Hughes, 1999)

Well	Number of Samples	Observation Period (yrs)	Exponent	Half-Life (yrs)	Correlation Coefficient
MW-1	5	1.3	-0.0025	0.76	0.39
MW-2	5	1.3	-0.0065	0.29	0.91
MW-3	5	1.3	-0.0039	0.49	0.93

TABLE 7

Radionuclide Half-Lives and Specific Activities
(from Friedlander et al., 1981)

Radionuclide	**Half-Life (years)**	**Specific Activity (Ci/g)**
Ra-226	1600	0.99
Ra-228	5.76	234
Th-228	1.91	821.2
Th-230	8×10^4	0.019
Th-232	1.41×10^{10}	1.09×10^{-7}
U-234	2.45×10^5	0.006
U-235	7.038×10^8	2.16×10^{-6}
U-238	4.5×10^9	3.34×10^{-7}

TABLE 8

Correlation Coefficients for Gross Alpha and Gross Beta versus Analyzed Radionuclides

Parameter	**Gross Alpha**	**Gross Beta**
Gross Alpha	1.00	0.40
Gross Beta	0.40	1.00
Potassium (elemental)	0.43	**0.99**
Calculated K-40	0.43	**0.99**
Ra-226	0.16	0.53
Ra-228	0.45	0.11
Total Radium	0.44	0.47
U-234	**0.61**	0.32
U-235	-0.06	-0.22
U-238	**0.53**	0.19
Total Uranium	**0.60**	0.27
Th-228	0.14	0.00
Th-230	0.17	0.02
Th-232	0.04	-0.02
Total Thorium	0.22	0.01

TABLE 9

Summary Statistics for Radiological Parameters

Parameter	Count	% ND	Minimum (pCi/L)	Median (pCi/L)	Mean (pCi/L)	Maximum (pCi/L)
Gross Alpha	54	39	0.00	4.80	8.42	67.74
Gross Beta	53	17	0.02	8.07	52.96	711.00
Calculated K-40	53	0	0.62	5.20	45.46	593.49
Ra-226	54	26	-2.30	0.39	0.45	2.28
Ra-228	54	74	0.05	0.80	0.75	2.08
Total Radium	54		0.12	1.17	1.25	3.34
U-234	54	6	0.00	1.19	2.66	11.01
U-235	54	91	-0.03	0.18	0.23	0.70
U-238	54	26	0.00	0.74	1.34	5.01
Total Uranium	54		0.25	2.14	4.22	15.81
Th-228	54	80	0.00	0.23	0.27	0.81
Th-230	54	28	0.00	0.55	0.58	2.01
Th-232	54	72	0.00	0.21	0.23	0.63
Total Thorium	53		0.43	1.07	1.10	2.85

ND = Not Detected

TABLE 10

Adjusted Gross Beta Exceedances

Well S&W Lab Sample ID Screened Interval (ft bgs)	MW-13D 05a-021542 25'-50'		BRPZ-2RE 12b-021607 44'-64'		BRPZ-5 12b-021576 45'-55'		MW-26D 12b-021642 35'-60'	
	Result	**Uncertainty**	**Result**	**Uncertainty**	**Result**	**Uncertainty**	**Result**	**Uncertainty**
Gross Beta β (pCi/L)	84.75	± 3.56	473.2 J	± 39.98	205.98	± 49.22	711.0	± 84.0
Potassium (µg/L)	21,600	± 4,320	418,000 J	± 83,600	209,000 J	± 41,800	813,000	± 162,600
Calculated K-40 (pCi/L)	15.77	± 3.15	305.14	± 61.03	152.57	± 30.51	593.49	± 118.70
Adjusted Gross β (pCi/L)	68.98	± 4.76	168.02	± 72.96	53.41	± 57.91	117.51	± 145.41

Table 11

Adjusted Gross Alpha Exceedances

Well Screened Interval (ft bgs)	MW-3D 30'-55' Result	 Uncertainty	BRPZ-2RE 44'-64' Result	 Uncertainty	BRPZ-5 69'-79' Result	 Uncertainty
Gross Alpha (pCi/L)	18.93 UJ	± 17.19	33.96	± 25.03	67.74	± 20.25
U-234 Result (pCi/L)	0.126 U	± 0.24	5.10	± 1.58	6.75	± 1.86
U-235 Result (pCi/L)	0.38 U	± 0.04	0.075 U	± 0.18	0.177 U	± 0.29
U-238 Result (pCi/L)	0.00 U	± 0.19	1.78	± 0.81	3.33	± 1.14
Adjusted Gross Alpha (pCi/L)	18.42 U	± 17.2	27.00	± 25.10	57.47	± 20.37

Figure 1. Potassium-40 vs. Gross Beta Activities in Bedrock Groundwater Samples

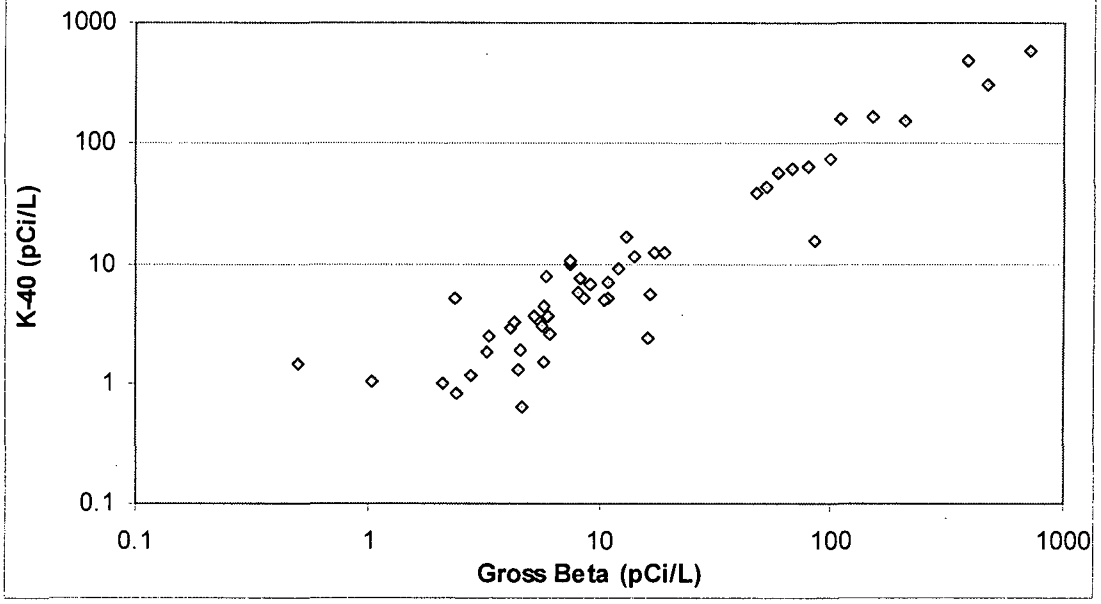


Figure 2. Potassium-40 vs. Gross Beta Activities in Bedrock Groundwater Samples (with uncertainties)

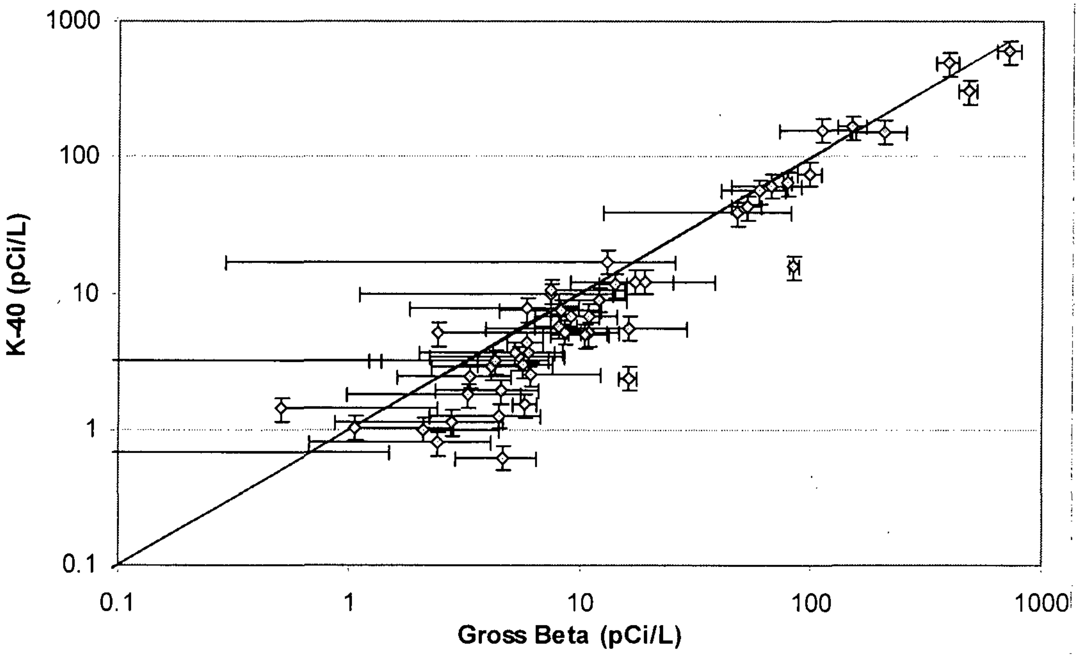
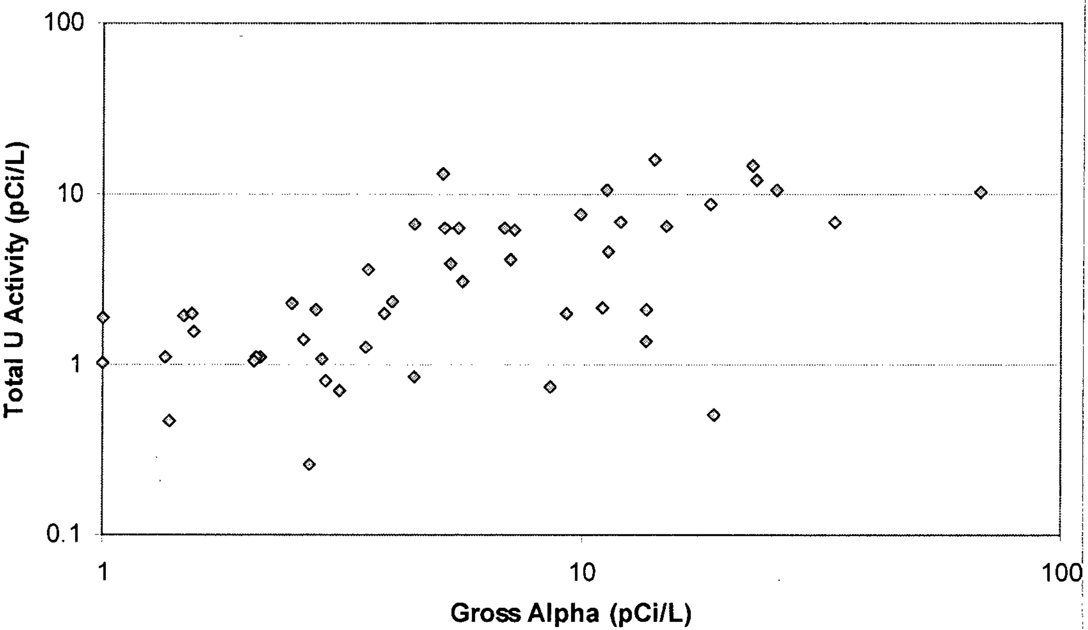


Figure 3. Total Uranium vs. Gross Alpha Activity



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APPENDIX C

GROUNDWATER FLOW AND SOLUTE TRANSPORT MODELING RESULTS (VOLUME 2)

APPENDIX D

DETAILED COST SUMMARIES
(VOLUME 1)

Appendix D
Table 1
Summary of Costs
Maywood GWFS
Maywood, New Jersey

ALTERNATIVE NUMBER	CAPITAL COSTS	O & M TOTAL	PRESENT WORTH [1]
Alternative 1 - No Action	$0	$0	$0
Alternative 2 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, and Non-Radiological Soil Remediation	$10,332,000	$20,122,000	$30,454,000
Alternative 3 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, In Situ Treatment with Redox Alteration, and Non-Radiological Soil Remediation	$14,482,000	$21,447,000	$35,929,000
Alternative 4 - Use Restrictions, Groundwater Monitoring, Groundwater Extraction with Ex Situ Treatment, Groundwater Discharge, and Non-Radiological Soil Remediation	$12,936,000	$109,266,000	$122,202,000

NOTES:

1. The 30 year present worth costs are for capital and O&M. Present worth calculated based on 2.7% annual discount rate.

Appendix D
Table 2
Alternative 1 - No Action
Capital Costs and Cost Summary
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity	Units	Unit Cost	Item Cost
General Remedial Activities				
No Actions Taken				$0
Remedial Action Costs Subtotal				**$0**
Overhead/QA Costs	30%			$0
Contingency Costs	20%			$0
Remediation Costs Subtotal				**$0**
Lifetime O&M Costs (Present Worth)	NA			**$0**

Total Present Worth for Alternative 1	**$0**

Appendix D
Table 3
Alternative 1 - No Action
Operation and Maintenance Costs
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity Per Year	Units	Unit Cost	Annual Cost	Number of Yearly Events	Present Worth Cost
Not Applicable						
No Actions Taken				$0		$0

Lifetime O&M Costs (Present Worth)	**$0**

Appendix D
Table 4

Alternative 2 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, and Non-Radiological Soil Remediation

Capital Costs and Cost Summary
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity	Units	Unit Cost	Item Cost
General Remedial Activities				
Project QA / QC Plan	1	LS	$50,000	$50,000
Project O&M Plan	1	LS	$50,000	$50,000
Natural Attenuation/Groundwater Monitoring Work Plan	1	LS	$50,000	$50,000
Professional Surveying	1	LS	$20,000	$20,000
Cleanup and Demobilization	1	LS	$15,000	$15,000
Classification Exception Area (CEA)	1	LS	$19,400	$19,400
CEA Deed Notice	20	Each	$696	$13,900
Notification of Property Owners (registered mail)	20	Each	$116	$2,300
Non-Radiological Soil Remediation				
Above Water Table	8000	CY	$500	$4,000,000
Below Water Table	2000	CY	$900	$1,800,000
Project Management				$602,100
Remedial Action Costs Subtotal				**$6,623,000**
Overhead/QA Costs	30%			$1,986,900
Contingency Costs	20%			$1,722,000
Remediation Costs Subtotal				**$10,332,000**
Lifetime O&M Costs (Present Worth)	From Appendix D Table 5			**$20,122,000**
Total Present Worth for Alternative 2				**$30,454,000**

Notes:

1. Site closeout activities are presented on the accompanying O & M cost table.
2. Soil remediation includes all labor, materials, equipment, testing, and disposal costs. Excavation will occur above and below the water table. Water treatment is included for excavations below the water table.

Appendix D
Table 5

Alternative 2 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, and Non-Radiological Soil Remediation

Operation and Maintenance Costs
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity Per Year	Units	Unit Cost	Annual Cost	Number of Yearly Events	Present Worth Cost
Quarterly Monitoring (Years 1 and 2)						
Well installation, Well Rehab, Sampling, Lab Analysis, and Validation	1	LS	$380,696	$380,696	1	$595,600
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$296,696	$296,696	1	$451,900
Annual Monitoring (Years 3 through 13)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$190,000	$190,000	11	$2,796,400
Annual Monitoring (Years 14 through 30)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$168,100	$168,100	17	$2,646,300
Monitoring Report						
Annual Monitoring Report	1	Each	$60,000	$60,000	30	$2,017,900
Year Two Report - from Quarterly Monitoring Data	1	Each	$60,000	$60,000	1	$93,900
Five Year Report - CERCLA Review	1	Each	$60,000	$60,000	6	$382,400
Non Radiological Soil Remediation (Years 2 and 3)						
Above Water Table (Year 2)	7600	CY	$500	$3,800,000	1	$5,944,700
Below Water Table (Year 2)	3400	CY	$900	$3,060,000	1	$4,787,000
Above Water Table (Year 3)	0	CY	$500	$0	1	$0
Below Water Table (Year 3)	0	CY	$900	$0	1	$0
Site Closeout (Year 3)						
Abandonment of Monitoring Wells - Bedrock Wells	49	Each	$2,500	$122,500	1	$186,600
Abandonment of Monitoring Wells - Overburden Wells	34	Each	$2,500	$85,000	1	$129,500
Site Closeout (Year 30)						
Abandonment of Monitoring Wells - Bedrock Wells	24	Each	$2,500	$60,000	1	$44,500
Abandonment of Monitoring Wells - Overburden Wells	24	Each	$2,500	$60,000	1	$44,500

Lifetime O&M Costs (Present Worth)	**$20,122,000**

Notes:

1. O&M costs are totaled as a present worth cost based on a 2.7% net investment rate for the period of time noted.
2. Lifetime O&M costs include annual QA and contingency costs.
3. The O&M costs include 25% for overhead / QA costs, 20% for O&M contingency costs, and 10% for project management.

Appendix D
Table 6
Alternative 3 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, In Situ Treatment with Redox Alteration, and Non-Radiological Soil Remediation
Capital Costs and Cost Summary
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity	Units	Unit Cost	Item Cost
General Remedial Activities				
Project QA / QC Plan	1	LS	$50,000	$50,000
Project O&M Plan	1	LS	$50,000	$50,000
In Situ Groundwater Monitoring Work Plan	1	LS	$50,000	$50,000
Utility Clearance and Injection Point Layout	1	LS	$25,000	$25,000
Professional Surveying	1	LS	$20,000	$20,000
Cleanup and Demobilization	1	LS	$15,000	$15,000
Construction Reports, Completion Reports, As-Built Drawings	1	LS	$30,000	$30,000
Classification Exception Area (CEA)	1	LS	$19,400	$19,400
CEA Deed Notice	20	Each	$696	$13,900
Notification of Property Owners (registered mail)	20	Each	$116	$2,300
Non-Radiological Soil Remediation				
Above Water Table	8000	CY	$500	$4,000,000
Below Water Table	2000	CY	$900	$1,800,000
In Situ Treatment				
Bench-Scale Study, Health and Safety Plan, Materials List, and Procurement	1	LS	$188,400	$188,400
Site Preparation	1	LS	$25,100	$25,100
Design	1	LS	$60,000	$60,000
Treatment Injection for Bedrock Areas	1	LS	$0	$0
Treatment Injection for Overburden Areas	1	LS	$2,090,000	$2,090,000
Project Management				$843,900
Remedial Action Costs Subtotal				**$9,283,000**
Additional Costs - % of Remedial Costs				
Overhead/QA Costs	30%			$2,784,900
Contingency Costs	20%			$2,413,600
Remediation Costs Subtotal				**$14,482,000**
Lifetime O&M Costs (Present Worth)	From Appendix D Table 7			**$21,447,000**
Total Present Worth for Alternative 3				**$35,929,000**

Notes:

1. Site closeout activities are presented on the accompanying O & M cost table.
2. Soil remediation includes all labor, materials, equipment, testing, and disposal costs. Excavation will occur above and below the water table. Water treatment is included for excavations below the water table.

Appendix D
Table 7
Alternative 3 - Use Restrictions, Groundwater Monitoring, Monitored Natural Attenuation, In Situ Treatment with Redox Alteration, and Non-Radiological Soil Remediation
Operation and Maintenance Costs
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity Per Year	Units	Unit Cost	Annual Cost	Number of Yearly Events	Present Worth Cost
In Situ Treatment						
Follow-up Treatment, Year 2 (30% of Year 1 costs)	1	LS	$645,000	$645,000	1	$1,009,000
Follow-up Treatment, Year 3 (10% of Year 1 costs)	1	LS	$215,000	$215,000	1	$327,500
Quarterly Monitoring (Years 1 and 2)						
Well installation, Well Rehab, Sampling, Lab Analysis, and Validation	1	LS	$380,696	$380,696	1	$595,600
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$296,696	$296,696	1	$451,900
Annual Monitoring (Years 3 through 13)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$190,000	$190,000	11	$2,796,400
Annual Monitoring (Years 14 through 30)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$167,400	$167,400	17	$2,635,300
Monitoring Report						
Annual Monitoring Report	1	Each	$60,000	$60,000	30	$2,017,900
Year Two Report - from Quarterly Monitoring Data	1	Each	$60,000	$60,000	1	$93,900
Five Year Report - CERCLA Review	1	Each	$60,000	$60,000	6	$382,400
Non Radiological Soil Remediation (Years 2 and 3)						
Above Water Table (Year 2)	7600	CY	$500	$3,800,000	1	$5,944,700
Below Water Table (Year 2)	3400	CY	$900	$3,060,000	1	$4,787,000
Above Water Table (Year 3)	0	CY	$500	$0	1	$0
Below Water Table (Year 3)	0	CY	$900	$0	1	$0
Site Closeout (Year 3)						
Abandonment of Monitoring Wells - Bedrock Wells	49	Each	$2,500	$122,500	1	$186,600
Abandonment of Monitoring Wells - Overburden Wells	34	Each	$2,500	$85,000	1	$129,500
Site Closeout (Year 30)						
Abandonment of Monitoring Wells - Bedrock Wells	24	Each	$2,500	$60,000	1	$44,500
Abandonment of Monitoring Wells - Overburden Wells	24	Each	$2,500	$60,000	1	$44,500

Lifetime O&M Costs (Present Worth)	**$21,447,000**

Notes:

1. O&M costs are totaled as a present worth cost based on a 2.7% net investment rate for the period of time noted.
2. Lifetime O&M costs include annual QA and contingency costs.
3. The O&M costs include 25% for overhead / QA costs, 20% for O&M contingency costs, and 10% for project management.

Appendix D
Table 8
Alternative 4 - Use Restrictions, Groundwater Monitoring, Groundwater Extraction with Ex Situ Treatment, Groundwater Discharge, and Non-Radiological Soil Remediation
Capital Costs and Cost Summary
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity	Units	Unit Cost	Item Cost
General Remedial Activities				
Project QA / QC Plan	1	LS	$50,000	$50,000
Project O&M Plan	1	LS	$50,000	$50,000
Groundwater Monitoring Work Plan	1	LS	$50,000	$50,000
Professional Surveying	1	LS	$20,000	$20,000
Cleanup and Demobilization	1	LS	$15,000	$15,000
Construction Reports, Completion Reports, As-Built Drawings	1	LS	$30,000	$30,000
Classification Exception Area (CEA)	1	LS	$19,400	$19,400
CEA Deed Notice	20	Each	$696	$13,900
Notification of Property Owners (registered mail)	20	Each	$116	$2,300
Non-Radiological Soil Remediation				
Above Water Table	8000	CY	$500	$4,000,000
Below Water Table	2000	CY	$900	$1,800,000
Groundwater Extraction and Treatment System				
System Design	1	LS	$115,927	$115,900
Bench-Scale Study, Health and Safety Plan, Materials List, and Procurement	1	LS	$218,407	$218,400
Site Preparation	1	LF	$29,098	$29,100
Extraction Well Installation	6	Each	$20,000	$120,000
Piping from Wells to Central Collection Line (trenching, backfill, compaction)	600	LF	$131	$78,600
Piping from Central Collection to Treatment Plant (trenching, backfill, compaction)	1,200	LF	$131	$157,200
Electric Supply Lines to Wells	1,800	LF	$29	$52,200
Electric Pumps for Extraction Wells	6	Each	$3,721	$22,300
Treatment Plant Building Construction, 30' x 30'	1	LS	$142,000	$142,000
Air stripper	1	LS	$18,637	$18,600
Initial Carbon for air stream	1	LS	$6,258	$6,300
Transfer pumps	1	LS	$2,481	$2,500
Metals Precipitation Equipment	1	LS	$148,851	$148,900
RO / Ion Exchange unit	1	LS	$155,053	$155,100
Piping inside Treatment Plant	1	LS	$18,606	$18,600
Setup and installation of equipment - labor	1	LS	$48,000	$48,000
Piping from Treatment to POTW (trenching, backfill, compaction)	2,500	LF	$47	$117,500
Permit for Discharge to POTW	1	LS	17,389	$17,400
System Startup	1	LS	18,548	$18,500
Project Management				$753,800
Remedial Action Costs Subtotal				**$8,292,000**
Additional Costs - % of Remedial Costs				
Overhead/QA Costs	30%			$2,487,600
Contingency Costs	20%			$2,155,900
Remediation Costs Subtotal				**$12,936,000**
Lifetime O&M Costs (Present Worth)	From Appendix D Table 9			**$109,266,000**
Total Present Worth for Alternative 4				**$122,202,000**

Notes:
1. Site closeout activities are presented on the accompanying O & M cost table.
2. Soil remediation includes all labor, materials, equipment, testing, and disposal costs. Excavation will occur above and below the water table. Water treatment is included for excavations below the water table.

Appendix D
Table 9
Alternative 4 - Use Restrictions, Groundwater Monitoring, Groundwater Extraction with Ex Situ Treatment, Groundwater Discharge, and Non-Radiological Soil Remediation
Operation and Maintenance Costs
Maywood GWFS
Maywood, New Jersey

Item Description	Quantity Per Year	Units	Unit Cost	Annual Cost	Number of Yearly Events	Present Worth Cost
Quarterly Monitoring (Years 1 and 2)						
Well installation, Well Rehab, Sampling, Lab Analysis, and Validation	1	LS	$380,696	$380,696	1	$595,600
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$296,696	$296,696	1	$451,900
Annual Monitoring (Years 3 through 13)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$190,000	$190,000	11	$2,796,400
Annual Monitoring (Years 14 through 30)						
Well Rehab/Replacement, Sampling, Analysis, and Validation	1	LS	$168,100	$168,100	17	$2,646,300
Groundwater Operational Costs						
Well Cleaning	1	annual	$6,956	$6,956	30	$233,900
Discharge from New Extraction Wells and Treatment System (see Note 5)	5,256,000	Gal	$0.50	$2,628,000	30	$88,384,200
Replace extraction wells with possible relocation to optimize system	1	Each	$23,185	$23,185	See Note 3	$130,600
Pump Replacement	2	Each	$3,210	$6,420	See Note 3	$36,200
Additional Piping and Trenching	300	LF	$131	$39,300	See Note 3	$221,300
Additional Redevelopment of Wells	6	Each	$3,478	$20,867	See Note 3	$117,500
Monitoring Report						
Annual Monitoring Report	1	Each	$60,000	$60,000	30	$2,017,900
Year Two Report - from Quarterly Monitoring Data	1	Each	$60,000	$60,000	1	$93,900
Five Year Report - CERCLA Review	1	Each	$60,000	$60,000	6	$382,400
Non Radiological Soil Remediation (Years 2 and 3)						
Above Water Table (Year 2)	7600	CY	$500	$3,800,000	1	$5,944,700
Below Water Table (Year 2)	3400	CY	$900	$3,060,000	1	$4,787,000
Above Water Table (Year 3)	0	CY	$500	$0	1	$0
Below Water Table (Year 3)	0	CY	$900	$0	1	$0
Site Closeout (Year 3)						
Abandonment of Monitoring Wells - Bedrock Wells	49	Each	$2,500	$122,500	1	$186,600
Abandonment of Monitoring Wells - Overburden Wells	34	Each	$2,500	$85,000	1	$129,500
Site Closeout (Year 30)						
Abandonment of Extraction Wells	11	Each	$2,500	$27,500	1	$20,400
Abandonment of Monitoring Wells - Bedrock Wells	24	Each	$2,500	$60,000	1	$44,500
Abandonment of Monitoring Wells - Overburden Wells	24	Each	$2,500	$60,000	1	$44,500

Lifetime O&M Costs (Present Worth)	**$109,266,000**

Notes:

1. O&M costs are totaled as a present worth cost based on a 2.7% net investment rate for the period of time noted.
2. Lifetime O&M costs include annual QA and contingency costs.
3. Occurs in Years 5, 10, 15, 20, and 25.
4. The O&M costs include 25% for overhead / QA costs, 20% for O&M contingency costs, and 10% for project management.
5. Includes all utility, chemical, materials, labor, analytical, POTW permit, and disposal costs associated with operating the water treatment system.