

TECHNICAL MEMORANDUM

POSSIBLE EFFECTS OF ARSENIC AND HEAVY METAL IN SEDIMENT AND SURFACE WATER ON FUTURE POTABLE WATER SUPPLY DEVELOPMENT IN THE CENTRAL AREA AQUIFER AT THE WELLS G & H SUPERFUND SITE

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January 2005

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EXECUTIVE SUMMARY

The primary purpose of this Technical Memorandum is to evaluate the potential for inorganic contaminants, particularly arsenic in the Aberjona River surface water sediment, and wetland sediments to impact the prospects for potable water development in the Wells G and H Central Area Aquifer. Arsenic is the focus of this evaluation, although the potential impacts of other metals including chromium, lead, iron, and manganese are also considered. The probable fate of copper and mercury are also discussed. The influence of geochemical parameters such as oxidation reduction potential (ORP) and dissolved oxygen (DO) are also evaluated. This study is based on relevant technical papers, site-specific reports and existing data available for the Wells G and H site

The impacts posed by arsenic upon potable water development in the Wells G and H Central Area aquifer depend upon the availability of sources of arsenic and the potential for contaminant and transport to the wells. Arsenic in the river, wetland sediments, and groundwater poses a *potential* threat to potable water in addition to the threat posed by chlorinated solvents like trichloroethene (TCE). Whether arsenic poses an *actual* threat (i.e., whether it might be detected at levels greater than drinking water standards) depends on the geochemical and hydrologic properties of the aquifer, and the location and construction details of the water supply well.

Numerous reports and technical papers were reviewed for this evaluation of the relationship between the groundwater in the Central Area Aquifer, the metal and metalloid contaminants in the sediments of the Aberjona River, and its associated wetlands. Arsenic was the primary contaminant considered, because of its ubiquitous presence in the sediments of the river and wetlands, its toxicity, and its potential mobility in the environment.

The preponderance of evidence from this evaluation suggests that the arsenic and other metals in the sediments would not adversely affect the development of large-capacity potable water supply wells in the aquifer. The evidence that led to this conclusion includes the following:

- **Aquifer Geochemical Conditions** – Aquifer geochemical conditions (i.e., shallow aquifer with mildly reducing conditions and medium/deep aquifer with more oxidizing conditions) and the presence of ferric hydroxide solids in the overburden (including beneath the peat) suggest the aquifer does not support the mobilization of dissolved arsenic under pumping conditions.
- **Low Arsenic Concentrations in Oxic Surface Water** – Aberjona River surface water is oxidized and contains low levels of dissolved arsenic (up to 6 ug/L). Under aquifer pumping conditions, the induced infiltration of surface water to the aquifer combined with the presence of ferric hydroxide solids in the overburden and the retardation properties of the aquifer, would limit the mobilization of dissolved arsenic under pumping conditions.
- **Low Hydraulic Conductivity of Peat Deposits** – Sediments/peat containing elevated levels of metals, particularly arsenic, have a very low hydraulic conductivity compared with the remaining overburden aquifer. Near Well H, Zeeb (1996) estimated that the peat

layer contributed less than 4-percent of the production water (with 96-percent originating from other overburden groundwater and river surface water). The low conductivity peat source and the above-described geochemical conditions are not conducive to the mobilization of dissolved arsenic in the aquifer under pumping conditions.

- **Artificially Elevated Metals Concentrations in Groundwater** – Historical groundwater data sampled by non-low flow sampling techniques appear to have artificially elevated metals concentrations compared to more recently data collected by EPA using low-flow techniques.
- **Historic Wells G and H Arsenic/Water Quality Data** – The historical metals data collected from former production wells G and H do not support the presence of arsenic above the current MCL (10 ug/L). However, the means and methods of sampling conducted at wells G and H are not documented for the available data; therefore, the potential influence of sampling technique on monitoring results cannot be determined.

Collectively, the above-summarized information are the major lines of evidence suggesting limited potential to mobilize dissolved arsenic to a production well under pumping conditions.

Review of the technical literature related to the geochemical behavior of arsenic provided the most compelling support for limited arsenic migration to the deep parts of the aquifer. Arsenic mobility in a groundwater system is strongly affected by the amount of iron in the system and the oxidation state of iron and arsenic. Arsenic, particularly arsenate (As[V]), strongly sorbs to ferric iron hydroxide solids under oxidizing conditions. Recent water quality data from various monitoring wells in the Central Area aquifer indicate that at greater depths in the aquifer, conditions become less reducing than they are in the shallow aquifer where the organic material in the overlying peat promotes reducing conditions, which can mobilize arsenic. The less reducing conditions are due to the effects of both less organic matter at greater depths in the aquifer and infiltration of oxic/less reduced water. As the induced infiltration passes into less reducing conditions below the peat, any advectively-transported arsenic could be sorbed to the ample ferric hydroxide solids and rendered immobile. Iron concentrations in excess of the Secondary Maximum Contaminant Level (SMCL), 0.3 mg/L, are widespread throughout the Central Area Aquifer. The hypothesized presence of ample iron oxyhydroxide solids is based on the generally high levels of aqueous iron (> SMCL) and high soil/overburden levels of iron at the nearby Olympia site recently investigated by EPA, which frequently exceeded percent levels for iron. Iron concentrations in soil at the Olympia Site ranged from 6,500 J mg/kg to 14,000 J mg/kg at a 100-percent rate of detection. In another hypothesized process, reduced iron transported from shallow portions of the aquifer could potentially undergo oxidation and precipitation resulting in capture of concurrently transported arsenic. In addition, under certain reducing conditions, which appear to be present in the peat sediments, the arsenic would be coprecipitated in the peat with the formation of iron sulfide and pyrite, thus limiting arsenic mobility.

Based on the pH and oxidation-reduction (redox) conditions (Eh) in the Central Aquifer area, a significant fraction of the arsenic will exist as arsenate, As(V), anions, which adsorbs more strongly to ferric iron hydroxide solids. Also, within this pH-Eh regime, a fraction of the total

iron should exist as ferric, Fe(III), iron solids. Detected total iron concentrations in groundwater are nearly three orders of magnitude greater than those of total arsenic (~10,000 ug/L versus ~10 ug/L), and the considerable presence of iron should provide more than ample ferric hydroxide surface area to control the solubility of arsenic in the Central Area aquifer.

Additional support that previous pumping did not transport arsenic can be inferred by seven parameters denoted by Smedley and Kinniburgh (2004) that typically accompany groundwater contaminated by arsenic. As summarized in Table ES-1, the parameters evaluated included iron, manganese, alkalinity, chloride, sulfate, nitrate, and pH based on monitoring data from several decades, including water quality data collected by the City of Woburn during the operation of municipal wells G and H as a water supply.

Analyte	Favored Range for High Arsenic (mg/L) ¹	Well G			Well H			Finding	
		Avg	Low	High	Avg	Low	High	Well G	Well H
Iron	>0.2	0.045	0.000 ²	0.210	0.631	0.010	2.400	Unfavorable	Favorable
Manganese	>0.5	0.410	0.020	1.170	1.123	0.200	2.160	Unfavorable	Favorable
Alkalinity	>500	50	32	60	50	47	56	Unfavorable	Unfavorable
Chloride	<60	69	19	185	90	25	116	Unfavorable	Unfavorable
Sulfate	<1	89	22	143	114	80	150	Unfavorable	Unfavorable
Nitrate	<1	3	0.4	21	3.8	0.03	32	Unfavorable	Unfavorable
pH	>7 SU	6.6	6.2	7.3	6.5	6.3	6.7	Unfavorable	Unfavorable

Notes:

¹ From Smedley and Kinniburgh, 2004.

² As posted in Mayor 2003.

SU – Standards units for potential of Hydrogen (pH).

Wells G and H data sources: Mayor, 2003; GeoTrans, 1994; and Dufresne-Henry, 1978

Evaluating the Wells G and H data in comparison to these parameters showed that unfavorable conditions were present in Well G (all 7 parameters unfavorable) and in Well H (5 of 7 parameters unfavorable) for the presence of high arsenic levels.

Under pumping conditions, some degree of induced surface water infiltration is expected. Aberjona River water is well oxygenated and contains low concentrations of dissolved arsenic (average baseflow concentration of approximately 6 ug/L as documented in TTNUS [2002]). Induced infiltration of surface water is expected to shift aquifer equilibrium along flow paths to more oxidizing conditions, which are favorable to the less mobile and more strongly sorbed As(V) form of arsenic. In addition, more oxidized conditions in the aquifer will favor the predominance of As(V) iron (III) oxyhydroxide solids, to which arsenate ions are very strongly sorbed. Longer term pumping is expected to favor this equilibrium shift over the long term.

The hydrologic relationship between the Central Area aquifer and the Aberjona River and wetlands was revealed to a large degree by a 30-day aquifer pumping test conducted at Wells G and H for EPA by the United States Geological Survey (USGS) in 1985/1986. By monitoring stream flow in the Aberjona River above and below the Well G and H area, the USGS showed

that after several hours of pumping, the river had switched from gaining to losing and significant induced infiltration of surface water was occurring. The rate of induced infiltration from the river eventually rose to about one-half of the pumping rate from the wells. The large flux of water through the Aberjona River sediments raises the potential concern that arsenic and other metals present may migrate to the supply wells. However, hydrogeologic conditions alone mitigate this potential migration concern. For example, the arsenic-contaminated wetland soils are actually a relatively thin layer compared to the entire aquifer thickness and contain a relatively small amount of groundwater. Once these deposits have been partially dewatered by the pumping, they are greatly diminished as a source of metals or metalloids. The bulk of the induced infiltration in the long-term is probably seepage through the riverbed. This seepage is likely not uniformly distributed along the river and may be greatest in non-depositional areas, where the thickness of sediment and peat is expected to be minimal. In 2002, EPA collected four-foot sediment core samples from 4 locations in the Wells G and H wetland as part of the Aberjona River Study (Operable Unit 3 [OU-3]). One of these four foot core samples encountered saturated sand before reaching 4 feet (approximately 2 to 3 feet) whereas the remaining core encountered peat and/or organic deposits as deep as 4 feet, thus documenting the variability in peat thickness/depth. Nonetheless, the opportunities for movement of the metals from the sediments into the aquifer may be significantly less than that implied by the rate of induced infiltration revealed by the USGS pumping test.

As demonstrated in Section 6.0, historical groundwater metals data collected using non-low flow techniques appear to be elevated compared to more recently collected low-flow groundwater data. The non-low flow sampling likely induced turbidity from the entrainment of aquifer solids. EPA's recently collected (2002) low-flow groundwater data are in many cases orders of magnitude lower in concentration than the older non-low-flow data. The older data are not representative of actual groundwater contaminant conditions and cast uncertainty on prior conclusions drawn by others using the non-low-flow data regarding "widespread" inorganic contamination in the Central Area Aquifer.

While municipal Wells G and H were not known to be tested for arsenic during their active operation as water supply wells, samples were collected and tested several months after the wells were taken off line in 1979. The 1979 samples had arsenic concentrations of 1.5 (Well H) and 2.0 (Well G) micrograms per liter (ug/L). Subsequent samples taken from Wells G and H, including one set taken during a 30-day pumping test in 1985/1986, showed arsenic concentrations to be less than the Environmental Protection Agency (EPA) 2006 Maximum Contaminant Limit (MCL) of 10 ug/L. The results suggest that arsenic in the river and sediments has not reached the deep part of the aquifer at concentrations high enough to result in levels above 10 ug/L in the former supply wells. In addition, concentrations in other pumping wells nearby (the former John J. Riley production wells) have been shown to not contain arsenic above the current EPA MCL of 10 ug/L. Note that the data collected from Wells G and H and the John J. Riley well were collected prior to the widespread use of EPA endorsed low-flow sampling techniques and could be biased high from aquifer solids entrainment. However, sampling records were not available from the parties who conducted the sampling to verify this hypothesis. In addition, the degree and/or impact of induced surface water infiltration is not known, which may dilute arsenic aquifer concentrations and cause incrementally greater partitioning of arsenic to solids that is favored under more oxic conditions.

Further indirect evidence that arsenic did not contaminate the municipal Wells G and H includes Woburn area hair sampling results, which indicate that there is no positive correlation between Wells G and H drinking water consumption and arsenic levels in hair samples (Rogers et al. 1997). However, there were likely several opportunities for direct or indirect arsenic removal from the water supply system prior to delivery to individual taps. Water conditioning or reactions during residence in the water distribution system could limit concentrations of arsenic consumed by individuals.

Collectively, this evidence suggests limited potential for the migration of arsenic to a supply well at or near Wells G and H above the MCL. The conclusion of this analysis is not an endorsement by the U.S. Environmental Protection Agency (EPA) or EPA's supporting contractors that treatment will not be required in the event that the Central Area aquifer is used as a future potable public water supply

1.0 INTRODUCTION

1.1 Purpose, Scope and Organization of the Report

Metcalf & Eddy (M&E) received Work Assignment (WA) No. 107-RICO-0146 under the United States Environmental Protection Agency (EPA) Response Action Contract (RAC) No. 68-W6-0042 to perform technical support at the Wells G and H Superfund Site Operable Unit 3 (OU-3), (Aberjona River) in Woburn, Massachusetts (i.e., the Site). M&E assigned primary responsibility for preparation of the Technical Memorandum under this project to RAC Team Subcontractor, TRC Environmental Corporation (TRC). Primary activities included the preparation of a Technical Memorandum to evaluate the potential impact of river and wetland sediment contaminants, particularly inorganic contaminants and metals of concern (e.g., arsenic) on the Wells G and H Central Area aquifer using existing chemical contaminant data.

The text of the report is presented in the following sections, consistent with the approved Work Plan:

- Section 1.0, Introduction, presents study objectives;
- Section 2.0, Background, presents issues associated with other contaminant sources in the Central Area;
- Section 3.0, Documentation, summarizes key technical documents used to prepare this Technical Memorandum;
- Section 4.0, Site Physical Characterization, presents site physical characteristics;
- Section 5.0, Relevant Chemical Behavior in the Environment, summarizes the fate and transport characteristics of metals of concern;
- Section 6.0, Analysis, contains the analysis of the available information;
- Section 7.0, Conclusions and Recommendations; and
- Section 8.0, References.

1.2 Project Objectives

The primary purpose of this Technical Memorandum is to evaluate the potential impact of river and wetland sediment contaminants, particularly arsenic, on the prospects for potable well development in the Wells G and H Central Area Aquifer. Existing chemical contaminant data were used. The Technical Memorandum considers relevant information on the nature and extent of sediment and surface water inorganic and metal contamination of concern within the Wells G and H 38-acre wetland, and the impact of that contamination on the underlying aquifer under existing conditions. Lines of evidence regarding the potential for the sediment and surface water contamination to migrate to a production well and impact drinking water quality are also discussed.

The secondary purpose of the Technical Memorandum is to document the source, nature, and extent of inorganic and metal contaminants of concern in the Central Area and their potential impact on the river/wetland sediments and surface water based on existing data compiled by Potentially Responsible Parties (PRPs) and EPA. Data obtained from a recent regulatory file review performed at the Massachusetts Department of Environmental Protection (MADEP) Northeast Regional Office, as summarized in TTNUS (2003b), are also qualitatively utilized to identify potential source areas and to describe the general nature and extent of contamination.

The Technical Memorandum also compiles and evaluates available published data on the hydraulic properties of the sediment and peat layer in the Aberjona River and associated wetland near municipal Wells G and H. Contaminant retardation, dilution, and fate and transport phenomena that can influence the potential for inorganic and metal contaminants to contaminate a newly pumped public water supply well in the Central Area Aquifer are also discussed.

The Technical Memorandum represents partial fulfillment of the 1989 Wells G and H Superfund Site Record of Decision Section X.A.2.E objective to "evaluate the mobility of contaminants including semi-volatile organics and metals under ambient and pumping conditions." This Technical Memorandum does not substitute for any actual or perceived requirements under the Consent Decree.

2.0 BACKGROUND

The Wells G and H Site is located in Woburn, Massachusetts as depicted in Figure 1. Figure 1 also identifies key geographic features of the Wells G and H Site.

The Wells G and H Central Area aquifer (OU-2) is currently classified as a GW-1 Potentially Productive Aquifer (MADEP, 2004) and has been used in the past as a major potable water supply for the City of Woburn, as well as for industrial water supply. Most of the Central Area aquifer is within the Interim Wellhead Protection Area (IWPA) of wells G and H. Wells G and H were closed due to the presence of elevated levels of chlorinated solvents in 1979, and the site was subsequently listed as a federal Superfund site in 1982. EPA's Record of Decision (ROD) for the Wells G and H Site calls for the restoration of the aquifer to potable standards (EPA, 1989). The Wells G and H PRPs have maintained that an important consideration in determining the level of appropriate cleanup for the Central Area (OU-2) aquifer is whether the aquifer could be suitable as a future drinking water supply (GeoTrans, 1994). The PRPs assert that there are a variety of groundwater contaminants in the Central Area (OU-2) aquifer that currently exceed groundwater and drinking water cleanup standards (e.g., tetrachloroethene, trichloroethene, vinyl chloride, arsenic, beryllium, chromium, cadmium, lead, nitrate, benzene, toluene), and further assert that restoration of the aquifer is technically impracticable due to other contaminants attributed to other sources that exceed cleanup goals.

The PRPs have undertaken remedial activities at the Source Area (OU-1) properties (see Figure 2). The Central Area (OU-2) aquifer has partially been remediated through the control of Source Area (OU-1) contributions from PRP sites; however, the chlorinated solvents that previously traveled offsite from PRP sites continue to impact the Central Area (OU-2) aquifer. The PRPs are reluctant to address the lingering offsite chlorinated solvents for the following two primary reasons:

- 1) Widely distributed concentrations of chlorinated solvents and other contaminants make it problematic to attribute specific contamination to individual PRPs.
- 2) The chlorinated solvents are only one portion of a larger spectrum of contaminants that exceed drinking water standards.

Due to these two primary factors the PRP groups concluded "that the ROD objective to restore the Central Area aquifer to drinking water quality is technically impracticable and that additional investigations and evaluations directed toward that objective are not warranted" (GeoTrans, 1994). (Note that whether the first point could be addressed through the joint and several liability aspects of Superfund is not the focus of this evaluation.)

This evaluation focuses on the second point, which is the impact of the larger spectrum of contamination (in particular arsenic). For example, the PRPs have claimed that contaminants in Aberjona River surface water and sediments, and contaminated wetland sediments, may serve as a continuing source of contamination to the Central Area aquifer, especially under pumping conditions. Consequently, after clean up of the contaminants caused by the PRPs the groundwater will still not meet drinking water standards for potable water according to the PRPs.

On a parallel track, the EPA is currently determining risks associated with the Aberjona River water and sediment quality.

Metal and inorganic contamination, particularly arsenic, appears to be the greatest potential concern for contaminant migration from sediments to the aquifer under a future pumping condition due to their extent of contamination, potential mobility and toxicity. The mobility of arsenic and other metals in groundwater is complex and influenced by a number of factors, including, but not limited to, the oxidation-reduction (redox) potential of the sediments and aquifer, pH, hydraulic gradients, hydraulic conductivity, sulfur and iron levels, and total organic carbon content. In addition, there are a number of potential direct groundwater sources of metals and inorganic groundwater contamination that may impact the Aberjona River and the Central Area (OU-2) aquifer, such as the upstream Industri-Plex Superfund site, the Olympia property, the John J. Riley Tannery property, apple orchard/agricultural operations, pesticide applications, and herbicide applications along railroad rights of way (ROWS).

This Technical Memorandum evaluates the potential for anthropogenic inorganic contaminants, particularly arsenic, in the Aberjona River surface water and wetland sediments to impact future water supply wells. This effort is based on relevant technical papers, site-specific reports, and existing data for the Wells G and H site.

3.0 DOCUMENTATION

Information used to prepare this Technical Memorandum includes site-specific data and reports, and technical literature.

3.1 Site-Specific Documentation and Data

The following site-specific documentation and data set forth in the approved work plan were considered during the preparation of this Technical Memorandum:

- *Wells G&H Site Central Area Remedial Investigation, Phase 1A Report.* Prepared for: Beatrice Corporation, UniFirst Corporation, W.R. Grace & Co. – Conn. Prepared by: GeoTrans, Inc. and RETEC. February 14, 1994.
- *Piezcone Mapping, Groundwater Monitoring, and Flow Modeling in a Riverine Peatland: Implications for the Transport of Arsenic.* Submitted to the Department of Civil and Environmental Engineering in Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy in Civil and Environmental Engineering at the Massachusetts Institute of Technology. Prepared by Peter John Zeeb. August 1996
- Technical papers by lead author Helena Solo-Gabriele, Ph.D.:
 - Solo-Gabriele, H.M. and F.E. Perkins, *Watershed-Specific Model for Streamflow, Sediment, and Metal Transport*, Journal of Environmental Engineering, January 1997,
 - Solo-Gabriele, H.M. and F.E. Perkins, *Metal Transport within a Small Urbanized Watershed*, Journal of Irrigation and Drainage Engineering, March-April 1997.
 - Solo-Gabriele, H.M. and F.E. Perkins, *Streamflow and Suspended Sediment Transport in an Urban Environment*, Journal of Hydraulic Engineering, September 1997.
 - Solo-Gabriele, H.M., *Generation of Long-Term Record of Contaminant Transport*, Journal of Environmental Engineering, July 1998.
- Historical land use data as described in the following documents prepared by the Woburn Redevelopment Authority (WRA):
 - Wells G&H Superfund Redevelopment Initiative. Advisory Committee Information Package. April 24, 2002.
 - Wells G&H Superfund Redevelopment Initiative. Advisory Committee Information Package. June 5, 2002.
 - Wells G&H Superfund Redevelopment Initiative. Advisory Committee Information Package. September 4, 2002.

- TRC Central Area Groundwater Analytical Data. October 2002.
- *Data Summary Report for the Former Drum Disposal Area, Wells G&H Superfund Site, Operable Unit 1 – Olympia Property, Woburn, Massachusetts.* Prepared for the United States Environmental Protection Agency. Prepared by TRC Environmental Corporation. December 2002.
- *Baseline Human Health and Ecological Risk Assessment Report, Wells G&H Superfund Site, Aberjona River Study, Operable Unit 3, Woburn, Massachusetts.* Prepared for the United States Environmental Protection Agency. Prepared by Metcalf & Eddy, Inc., September 2004.
- TTNUS Aberjona River Base and Storm Flow Surface Water Data May 2001 – October 2002.
- *Draft Preliminary MSGRP Supplemental Report, Southern Area, Remedial Investigation/ Feasibility Study, Industri-Plex Site, Woburn, Massachusetts.* Prepared for the United States Environmental Protection Agency. Prepared by TetraTech NUS, Inc. June 2003.
- *Final Project Report, Natural Attenuation Study, Groundwater, Surface Water, Soil and Sediment Investigation, Industri-Plex Superfund Site, Woburn, Massachusetts.* Prepared by Robert Ford, United States Environmental Protection Agency, Office of Research and Development, National Risk Management Research Laboratory, Subsurface Protection and Remediation Division, Subsurface Remediation Branch. September 2, 2004.
- Historical groundwater data from the City of Woburn during the operation of municipal water supply wells G & H (various dates).
- Wells G & H source area monitoring data (contained in annual operation and maintenance reports, various dates).

A summary of each of these documents is provided below:

3.1.1 Wells G&H Site Central Area Remedial Investigation, Phase 1A Report (GeoTrans, 1994)

GeoTrans (1994) prepared the *Wells G&H Site Central Area Remedial Investigation, Phase 1A Report* (Phase IA) on behalf of W.R. Grace & Co. – Conn (Grace), UniFirst Corporation (UniFirst), and Beatrice Corporation (Beatrice) pursuant to the September 21, 1990 Consent Decree between the PRPs and EPA. The report presents a description and discussion regarding the Central Area (OU-2) of the Wells G and H Site. The GeoTrans (1994) conceptual model of the Aberjona River watershed describes the hydrogeologic conditions that affect groundwater flow and chemical transport in the Central Area (OU-2); the interactions of the Aberjona River and the Central Area aquifer; the nature and extent of contamination in the Central Area; and past, present, and likely future sources of contamination.

The Phase IA effort included the installation of 188 monitoring wells, the collection of 728 groundwater and surface water samples, and 492 groundwater level measurements. Groundwater monitoring data from the monitoring well network were collected over an approximately 23 year period. Some of the data are validated. Some of the metals groundwater data were collected using low-flow procedures; however, earlier data from the 1980s and some from the early 1990s were not collected by low-flow procedures. The suite of metals analysis, however, is limited and includes only arsenic, lead, and chromium. Other inorganic analytes include chloride, sulfate, and nitrate.

GeoTrans asserted that the hydraulic connection between the Central Area aquifer and the Aberjona River resulted in the induced infiltration of contaminated Aberjona River surface water into the underlying Central Area aquifer in response to pumping of Wells G and H. GeoTrans also stated that under current hydraulic conditions (no groundwater withdrawal by the municipal wells), there is a natural groundwater discharge of approximately 450 gallons per minute (gpm) from the Central Area aquifer to the Aberjona River and associated wetlands, which GeoTrans expects to result in a natural extraction, or natural flushing of contaminated groundwater from the Central Area aquifer.

GeoTrans noted Central Area aquifer exceedances of drinking water standards and guidelines for a variety of chemicals and compounds including inorganic and organic compounds such as arsenic, beryllium, chromium, cadmium, lead, nitrate, benzene, toluene, tetrachloroethene, trichloroethene, and vinyl chloride. GeoTrans ascribed the contamination to a wide variety of sources other than the sources identified in the ROD, and claimed that it was not possible to define or map individual contaminant plumes for any significant distance due to the pervasive and widespread contamination. Nonetheless, GeoTrans described indications of previously undetected additional contaminant sources in the watershed. GeoTrans concluded that restoration of the Central Area aquifer was technically impracticable and additional investigation and evaluation was unwarranted (GeoTrans, 1994).

3.1.2 Piezocone Mapping, Groundwater Monitoring, and Flow Modeling in a Riverine Peatland: Implications for the Transport of Arsenic (Zeeb, 1996)

Zeeb (1996) evaluated the influence of deposit-scale features of a contaminated riverine wetland on the interaction between surface water and wetland groundwater, with a focus on the fate of riverborne arsenic in the wetland.

Zeeb conducted detailed stratigraphic mapping of the wetland that separates the Aberjona River at its closest approach to Well H using a wetland piezocone penetrometer. The mapping revealed an ice-block depression filled with silt overlain by several distinct peat strata and sand layers. The post-glacial history follows a progression from fresh water lacustrine deposition, to woody swamp, to sedge meadow, to cattail marsh. Zeeb determined that the distribution of arsenic in the soil profile suggested that arsenic was deposited on the former wetland surface in a sedimentary form, some of which was transported to depth in groundwater.

Zeeb (1996) indicated that the arsenic contained in a small sand layer in the shallow peat is relatively mobile, but suggested that the arsenic contained in streambed sediments could serve as

a source of arsenic to the outside wetland if the layers were extensive enough. Zeeb postulated that in areas of the river where the peat layer was thin, river water could have supplied Wells G and H, but in his study area the amount of peat water that could drain to Wells G and H would have been small based on Zeeb's study results.

3.1.3 *Four Technical Papers by Lead Author Helena Solo-Gabriele, Ph.D. (Solo-Gabriele, 1997 a,b,c, 1998)*

The four technical papers were based on the data and findings from Solo-Gabriele (1995) and describe the use of hydrograph separation techniques to interpret metals data in the Aberjona watershed. The metals include iron, chromium, copper and the metalloid arsenic. The river system was modeled as a series of box models to evaluate macro-scale effects of stream flow components and associated sediment transport characteristics on metal transport. The bulk effects of various water sources were characterized by different response times, and the timing was based on hydrographs.

Three major flow components were identified and characterized (quick storm flow, slow storm flow, and long-term baseflow). The geographic distribution of metals inputs was examined, with metals concentrations highest in the upper reaches of the Aberjona River and an overall decrease in the downstream direction. Total metal concentrations were higher during storm conditions compared to low flow situations. During storm events, particulate metal concentrations increased significantly above typical concentrations observed during low flows. Concentrations for all metals exhibited a strong flushing effect with bursts of particulate metal transport at the initiation of storm flow conditions (Solo-Gabriele and Perkins, 1997 a, b, and c). Retrospective modeling of the river showed that 72-percent of the arsenic was transported during low flow conditions, as opposed to 31-percent of the chromium. Most chromium (80-percent) was transported in the particulate phase, whereas arsenic was roughly equally distributed between dissolved and particulate phases (Solo-Gabriele, 1998).

3.1.4 *Wells G and H Superfund Redevelopment Initiative Information Packages (WRA, 2002 a,b,c)*

In July 13, 2000, EPA announced that five Superfund sites in Massachusetts, including the Wells G&H Site, would receive federal grants to help the recipient community plan for the productive use of a toxic waste site. The EPA Superfund Redevelopment Initiative was intended as an aggressive planning tool to encourage community decisions on the future use of abandoned and contaminated properties.

The City of Woburn, through the WRA is employing the funding to develop a reuse plan to evaluate reasonable future uses for the municipal parcels and adjacent properties now affected by contamination from the Wells G and H site. The City of Woburn involves the community and other stakeholders in deciding how to reuse the property, including developing a land use plan and participating in city-sponsored and EPA public meetings.

The three Wells G and H Superfund Redevelopment Initiative Information Packages (WRA, 2002 a,b,c) compiled meeting minutes, newspaper articles, and working papers (e.g., marketing

analyses, oil and hazardous waste release site reviews). The information packages include anecdotal information on past land use, discussions of zoning issues, and maps and figures illustrating proposed plans for properties in the Wells G and H Superfund site.

3.1.5 TRC Central Area Well Data (TRC, 2002 b,c)

In October, 2002, TRC sampled eight previously installed groundwater monitoring wells located in the Central Area of the Wells G and H Superfund Site to collect current information on the groundwater quality in the eastern portion of the Aberjona Aquifer and adjacent properties. Prior to this, chemical contaminant data from this area were last collected in the early 1990s. The wells sampled were: S-63S, S-68D, S-82, S-85S, S-86S, S-87D, S-89S, and UC-11. Tetrachloroethene (PCE) was selected as a marker contaminant for the chlorinated volatile organic compound (VOC) plume in this area.

The wells were chosen to assess the orientation, magnitude and extent of the plume. Existing data showed the plume orientation was approximately southwest towards the Aberjona River at one time, but deflected more due south towards Wells G and H when they began pumping. TRC estimates that it likely took several years after pumping ceased for the plume to move off this alignment to its current position. However, once the wells were shut down, the plume appears to have realigned to a more southwest pattern.

All samples were analyzed for VOCs, SVOCs, metals, and cyanide. Two wells were analyzed for polychlorinated biphenyls (PCBs) and pesticides (TRC, 2002b). All samples were collected using EPA Region I low stress (low-flow) sampling protocols except for samples collected from well UC-11. Monitoring well UC-11 is a Waterloo-style multi-level well that requires special equipment for sample collection. The sample from UC-11 was collected by GeoTrans personnel for analysis by EPA. Arsenic was detected in seven out of eight wells. All arsenic detections were below the current EPA MCL of 10 micrograms per liter (ug/L) for arsenic.

3.1.6 Data Summary Report for the Former Drum Disposal Area, Wells G&H Superfund Site, Operable Unit 1 – Olympia Property, Woburn, Massachusetts (TRC, 2002a)

The TRC (2002a) Data Summary Report organized and evaluated data collected from the Pre-Design Investigation and Site Characterization at the Olympia Property Superfund Site in Woburn, Massachusetts. The evaluation summarized newly collected data, discussed trends, and compared the data to existing contaminant standards and guidelines and identified data gaps.

The Olympia Site is the location of a Former Drum Disposal Area (FDDA). Available documentation indicated 17 drums were disposed sometime prior to their discovery in 1979/1980. The primary objectives of this TRC study were to delineate the trichloroethene (TCE) plume, define the clay layer in the FDDA, define hydraulics of the aquifer and silt clay layer at the FDDA, update data on Central Area and Olympia Terminal groundwater contamination, verify FDDA surface soil contamination identified by others, and delineate residual contaminated soil in subsurface soil in the FDDA.

Arsenic was not identified as an FDDA contaminant in prior site reports; however, arsenic was detected in all FDDA soil samples and in all 62 monitoring wells sampled by TRC at concentrations in excess of one or more comparison criteria. The highest concentrations of arsenic in groundwater tended to be shallow and located in the central Aberjona River valley, including the Olympia Terminal property. In these areas, 13 wells exceeded the current EPA MCL of 10 ug/L for arsenic. In the FDDA one soil sample yielded an arsenic concentration greater than the MADEP background concentration of 20 milligrams per kilogram (mg/Kg) [MADEP, 2002].

3.1.7 Baseline Human Health and Ecological Risk Assessment Report (Aberjona River Study) [M&E, 2004]

M&E (2004) issued the Baseline Human Health and Ecological Risk Assessment for the Aberjona River Study Area. The baseline risk assessment for the Aberjona River Study Area focused on sediments and soils along six miles of the Aberjona River and wetlands from Route 128 in Woburn to the Mystic Lakes in Arlington and Medford. The study area was divided into six sections, or reaches along the river. Reach 1 contains the Wells G and H Superfund Site and associated 38-acre wetland. Reach 2 contains a former cranberry bog to the south. After the cranberry bog, the river continues to flow south as a well-defined river channel through Reaches 3, 4 and 5 prior to discharging into Reach 6, the Mystic Lakes (EPA, 2003a).

The M&E study was based on analysis of over 390 sediment and soil samples from 52 sampling stations along the study area. Additional sediment samples were collected from 12 stations outside the study area to provide background information for comparison. Surface water and fish samples were collected from inside and outside the study area. EPA conducted various studies to more accurately characterize potential risks along the study area (EPA, 2003a).

Arsenic was present in sediments throughout the study area. Other metals, including antimony, chromium, copper, lead, mercury and zinc, were also detected at elevated levels. The Wells G and H 38-acre wetland exhibited some of the highest concentrations of metals within the study area (EPA, 2003a).

M&E concluded that sediments may pose a current health risk to people using the study area in two exposure areas along the east side of the Wells G and H 38-acre wetland (near the former municipal Well H), and in the irrigation channels along the western side of the center of the 17-acre former cranberry bog located to the south of Salem Street. Six other exposure areas evaluated for potential risks along the former cranberry bog showed no health risk (EPA, 2003b).

The ecological risk assessment did not reveal a risk to fish or green heron within the study area. However, risks were widely observed from depositional sediments in the Wells G and H 38-acre wetland and in the 17-acre former cranberry bog. Two sediment locations in the Mystic Lakes indicated potential risks to benthic invertebrates. The risks were primarily due to exposure to metals contamination in sediments and/or vegetation growing in contaminated sediments.

3.1.8 TTNUS Aberjona River Base and Storm Flow Surface Water Data May 2001 – October 2002 (TTNUS, 2002)

TTNUS collected data over an 18-month period from May 2001 to October 2002. Data collected included precipitation measurements, stream flow, total suspended solids (TSS), metals (total and dissolved), and select physico-chemical parameters (i.e., temperature, specific conductivity, dissolved oxygen [DO], potential of hydrogen [pH], oxidation reduction potential [ORP], and turbidity). Metals analysis included arsenic, iron, chromium, copper, lead, and mercury.

TTNUS observed that TSS impacts metals transport in surface water, since spikes in metals concentrations were often associated with spikes in TSS. Unique TSS behavior was observed at Station 4 (the discharge control structure for the Hall's Brook Holding Area [HBHA] at Mishawum Road).

TTNUS observed that arsenic concentrations, as well as the concentrations of other metals, tended to be higher in the northern portion of the river and near the Industri-Plex Superfund Site, with the highest concentrations generally observed at Station 2 and Station 4. TTNUS estimated that 80 to 90-percent of the arsenic in the Aberjona River originates from north of Route 128 and near Industri-Plex. Total metal concentrations typically decreased downstream of the HBHA outfall, with the most significant reductions being observed in the particulate phase.

TTNUS observed iron as the highest concentration metal during baseflow, with the highest iron concentrations observed in the HBHA outfall. TTNUS observed an apparent direct relationship between arsenic and iron concentrations in both the total and dissolved phases, but found no clear relationship between total metals concentrations and river flow rate.

The following table summarizes average contaminant and water quality data ranges during base flow from the 18-month TTNUS sampling program from Station 4 (the HBHA outflow north of Wells G and H and Station 5 (at the Salem Street bridge south of Wells G and H). These stations are representative of river water passing through the Wells G and H Central Area wetland.

Base Flow River Contaminant/Water Quality Monitoring Data												
Station	Arsenic		Iron		Chromium		Copper		Lead		Mercury	
	Dis	Tot	Dis	Tot	Dis	Tot	Dis	Tot	Dis	Tot	Dis	Tot
4	5.95	37.1	487	5,270	1.07	11.5	2.4	17.4	0.80	7.87	0.05	0.08
5	6.04	20.3	550	2,430	1.33	7.2	1.77	5.36	0.85	3.45	0.05	0.06

Water Quality Data (units as indicated)						
Station	Water Temp (°C)	Specific Conductivity (µg/cm)	Dissolved Oxygen (mg/L)	pH (Standard Units)	ORP (mV)	Turbidity (NTUs)
4	2.6 to 25.0	388 to 959	3.2 to 11.8	6.7 to 7.28	+184 to +440	10 to 294
5	2.4 to 24.4	454 to 1,079	4.2 to 11.8	6.53 to 7.05	+170 to +485	3.9 to 434

Notes:

Dis = Dissolved (filtered)

Tot = Total (unfiltered)

µg/L = micrograms per liter

°C = degrees Celsius

mg/L = milligrams per liter

mV = millivolts

NTUs = nephelometric turbidity units

The above-tabulated data from the TTNUS study demonstrate that dissolved metals concentrations are generally low in the reach of the Aberjona River passing Wells G and H. In addition, the water quality data demonstrate that the Wells G and H reach of the river is in a constant state of oxidation and oxygen enrichment.

3.1.9 Draft Preliminary MSGRP Supplemental Report, Southern Area, Remedial Investigation/ Feasibility Study, Industri-Plex Site, Woburn, Massachusetts (TTNUS, 2003)

TTNUS (2003) prepared this report to support the EPA's Multiple Source Groundwater Response Plan (MSGRP) investigations prescribed under the ROD for the Industri-Plex Superfund Site. TTNUS researched existing information on other potential source areas and evaluated their potential to impact the surface water and sediment quality of the Aberjona River.

To describe local hydrogeology, TTNUS summarized the findings of others (e.g., United States Geological Survey [USGS]) that report the tendency for groundwater to discharge to the Aberjona River through highly transmissive and unconfined stratified drift aquifer materials. The stream bed of the Aberjona River was described as leaky. Between rain events, water is primarily supplied to the Aberjona River by groundwater. TTNUS also noted that groundwater flow towards the river is reversed when the river stage rapidly increases, although the reversal effect is temporary, localized, and does not involve large volumes of water. [Although not identified in the TTNUS report, a similar localized groundwater flow reversal was noted by The RETEC Group, Inc. (RETEC) at the Wildwood Conservation Corporation (Wildwood) and Aberjona Autoparts properties, which was attributed to periodic high water conditions in the Aberjona River. Beaver dams on the river were also cited by RETEC as a contributing factor to high water conditions near these properties (RETEC, 2003).]

TTNUS identified several potential historical sources of arsenic in proximity to municipal Wells G and H and the 38-acre wetland, including agricultural operations northeast of Well H, former orchards at 200 Wildwood Avenue and at 399 Washington Street, and a greenhouse on

Washington Street. The TTNUS review noted the presence of arsenic in soil at the Wildwood property (60.7 mg/kg), discovery of numerous containers of arsenic trioxide on the Olympia property, and the presence of tannery wastes at properties occupying parts of the former John J. Riley Tannery property. Some of the properties on the subdivided John J. Riley (Riley) Tannery parcel have achieved a Response Action Outcome (RAO) under the Massachusetts Contingency Plan (MCP; 310 CMR 40.0000); however, some still have tannery wastes in place that could serve as a continuing source of contamination, particularly chromium.

Potential non-point sources identified in the TTNUS report included sewer overflows, municipal pesticide usage (gypsy moth spraying), roadway runoff and runoff from developed areas, and fertilizer applications, and the use of monosodium methane arsenate (MSMA) as an herbicide along the Boston and Maine railroad tracks.

3.1.10 Final Project Report, Natural Attenuation Study, Groundwater, Surface Water, Soil and Sediment Investigation, Industri-Plex Superfund Site, Woburn, Massachusetts (Ford, 2004b)

Ford (2004b) provides an assessment of arsenic contamination within the Industri-Plex Superfund Site and Groundwater/Surface Water Investigation Plan (GSIP) Study Area. The purposes of this study were to:

1. Determine the migration mechanisms controlling arsenic transport at the Industri-Plex Superfund Site and the GSIP Study Area;
2. Provide an evaluation of the potential role of natural attenuation processes in mitigating arsenic transport from the Industri-Plex Superfund Site and GSIP Study Area; and
3. Provide guidance for determination of reasonable, cost effective treatment technologies for a river/wetland in an urban watershed. The information derived from this report will be incorporated into the Industri-Plex Superfund Site, Operable Unit 2 (OU-2), MSGRP, and Remedial Investigation/Feasibility Study (RI/FS).

Ford investigated the migration of arsenic from suspected source areas within the Industri-Plex Superfund Site into the HBHA. Three goals were addressed as part of this investigation:

1. Identification of the mobile form of arsenic in groundwater;
2. Identification of the processes controlling arsenic uptake onto HBHA sediments; and
3. Evaluation of the stability of arsenic associated with HBHA sediments. This information serves as a basis for identifying the effectiveness of natural attenuation within the HBHA to mitigate downgradient migration of arsenic into the Aberjona River and for evaluation of potential remedial alternatives.

Ford (2004b) also documented the collection of sediments in the HBHA wetland and Wells G and H 38 acre wetland, including one sample collected near Well H (sediment sample GHME12B). The sediment samples were collected to assess the distribution and stability of

sediment-associated arsenic downgradient of the HBHA Pond. Select geochemical parameters and dissolved metals were analyzed on sediment pore water isolated from the sediment samples. In addition, subsamples of dried sediment were subjected to oxic leaching tests to assess arsenic release under oxic conditions.

Ford found that sediment pore waters contained elevated concentrations of arsenic ranging from approximately 160 to 670 ug/L, which did not correlate directly with the sediment arsenic concentration for all sediments. Following removal of the major portion of pore water, the sediments were dried and then analyzed for total element concentrations.

During oxic leaching tests of the sediments, uncontaminated, oxygenated, water was allowed to equilibrate with the dried sediments and the water quality was monitored over time to determine arsenic concentration. Measurements of the oxidation-reduction potential showed that sediment suspensions become more oxidizing over time. In all cases, most of the leached arsenic had been released within a 30-minute reaction period. Sediment samples GHME12B and GH2801 from the Wells G and H wetland showed the greatest arsenic release with average leachable concentrations of 341 ug/L (standard deviation 27 ug/L) and 112 ug/L (standard deviation 42 ug/L) respectively over a 4-hour period. The arsenic mass released per unit weight of sediment was 24 µg/g (standard deviation 1 µg/g) and 3 µg/g (standard deviation 1 µg/g) for GHME12B and GH2801, respectively. Arsenic release was less for sediment samples GH2901 (from the Wells G and H wetland south of Well H) and HBHAW02B (from the HBHA) with concentrations of 23 ug/L and 12 ug/L, respectively.

The batch test may not represent a continuous flow system in which the desorbed arsenic is continuously eluted from the reaction zone. Also, the batch tests do not capture the conditions that may be induced by microbial activity that could occur under induced infiltration (Ford, 2005a).

3.1.11 Historical City of Woburn Data from Municipal Water Supply Wells G & H (Various Dates)

Historical City of Woburn analytical testing of water from municipal water supply wells G and H was obtained from data compiled by the Massachusetts Department of Environmental Protection (MADEP). None of the analytical data included arsenic. However, data for iron, manganese and other inorganic analytes were included. Organic chemical data, pumping rates, and some related operational data were also included.

3.1.12 Wells G & H Source Area Monitoring Data

Routine Wells G and H Source Area (OU-1) monitoring data were limited to VOCs. No metals data were provided from monitoring wells routinely monitored as part of Source Area (OU-1) remediation/compliance activities.

3.2 Additional Technical Literature

The following additional technical literature was considered in the preparation of this Technical Memorandum:

- *City of Woburn, Woburn, Massachusetts Report on East Side Water Problems – Wells G&H.* Prepared for Department of Public Works, Albert J. Wall, Superintendent. Prepared by Dufresne-Henry Engineering Corporation, North Springfield, Vermont. January 1978.
- Cherry, J.A., A.U. Shaikh, D.E. Tallman, and R.V. Nicholson. *Arsenic Species as an Indicator of Redox Conditions in Groundwater.* Journal of Hydrology, Volume 43, Pages 373 to 392, 1979.
- Pierce, M.L. and C.B. Moore. *Adsorption of Arsenite and Arsenate on Amorphous Iron Hydroxide.* Water Resources, Volume 16, Pages 1247 to 1253, 1982.
- Brannon, J.M. and W.H. Patrick, Jr., *Fixation, Transformation, and Mobilization of Arsenic in Sediments.* Environmental Science and Technology, Volume 21, Number 5, Pages 450 to 459, 1987.
- Moore, J.N., W.H. Ficklin, and C. Johns. *Partitioning of Arsenic and Metals in Reducing Sulfidic Sediments.* Environmental Science and Technology, Volume 22, Number 4, Pages 432 to 437, 1988.
- Masscheleyn, P.H., R.D. Delaune, and W.H. Patrick, Jr. *Arsenic and Selenium Chemistry as Affected by Sediment Redox Potential and pH.* Journal of Environmental Quality, Volume 20, Pages 522 to 527, 1991.
- Masscheleyn, P.H., R.D. Delaune, and W.H. Patrick, Jr. *Effect of Redox Potential and pH on Arsenic Speciation and Solubility in a Contaminated Soil.* Environmental Science and Technology, Volume 25, Number 8, Pages 1414 to 1419, 1991.
- Knox, M.L. *The Distribution and Depositional History of Metals in Surface Sediments of the Aberjona River Watershed.* Submitted to the Department of Civil and Environmental Engineering in Partial Fulfillment of the Requirements for the Degree of Master of Science in Civil Engineering at the Massachusetts Institute of Technology. September 1991.
- Aurilio, A.C. *Arsenic in the Aberjona Watershed.* Submitted to the Department of Civil and Environmental Engineering in Partial Fulfillment of the Requirements for the Degree of Master of Science in Civil Engineering at the Massachusetts Institute of Technology. September 1992.
- Bialon, J.L. *Characterization of the Physical and Engineering Properties of the Aberjona Wetland Sediments.* Submitted to the Department of Civil and Environmental

Engineering in partial fulfillment of the requirements for the degree of Masters of Science in Civil and Environmental Engineering at the Massachusetts Institute of Technology. May 1995.

- Ayotte, J.D., M.G. Nielsen, G.R. Robinson, Jr., and R.B. Moore. *Relation of Arsenic, Iron, and Manganese in Groundwater to Aquifer Type, Bedrock Lithochemistry, and Land Use in the New England Coastal Basins*. United States Geological Survey, Water-Resources Investigations Report 99-4162. 1999.
- Hinkle, S.R. and D.J. Polette, *Arsenic in Groundwater of the Willamette Basin, Oregon, Geochemistry of Arsenic*. United States Geological Survey. Water Resources Investigations Report 98-4205. 1999.
- Ravenscroft, P., J.M. McArthur, and B.A. Hoque, *Geochemical and Paleohydrological Controls on Pollution of Groundwater by Arsenic*. In: *Arsenic Exposure and Health Effects IV*. W.R. Chappell, C.O. Abernathy, and R. Calderon (Eds), Elsevier Science Ltd. 2001.
- Keon, N.E., C.H. Schwartz, D.J. Brabander, C. Harvey, and H.F. Hemond, *Validation of an Arsenic Sequential Extraction Method for Evaluating Mobility in Sediments*. *Environmental Science and Technology*, Volume 35, Number 13, Pages 2778 to 2784, 2001.
- Wilkin, R.T. and R. G. Ford, *Use of Hydrochloric Acid for Determining Solid-Phase Arsenic Partitioning in Sulfidic Sediments*. *Environmental Science and Technology*, Volume 36, Number 22, Pages 4921 to 4927, 2002.
- Wilkin, R.T., D. Wallshlager, and R.G. Ford, *Speciation of Arsenic in Sulfidic Waters*. *Geochemical Transactions*. Volume 4, Number 1, Pages 1 through 7. 2003.
- EPA Research Brief (APM 04-73) - *The Impact of Groundwater-Surface Water Interactions on Contaminant Transport at Contaminated Sites (Draft)*, Prepared by Robert Ford, ORD/NRMRL/GWERD. July 28, 2004.
- P.L. Smedley and D.G. Kinniburgh, British Geological Survey, *Source and Behavior of Arsenic in Natural Waters*. World Health Organization, Water Sanitation and Health website. (http://www.who.int/water_sanitation_health/dwq/arsenic/en/) Accessed 2004.

The scope and relevance of each of these documents is summarized below:

3.2.1 City of Woburn, Massachusetts, Report on East Side Water Problems – Wells G&H (Dufresne-Henry, 1978)

Dufresne-Henry (1978) investigated problems with “East Side Well Water” on behalf of the City of Woburn Department of Public Works, particularly complaints of objectionable odors and taste. The Dufresne-Henry investigation was initiated on March 1, 1976.

The report included details of early well history. For example, when the wells were first started, there was evidence of coliform contamination and the State Health Department required that the water be chlorinated. Manganese was also present in the groundwater and the addition of chlorine caused precipitation to occur in the distribution system, leading to stained laundry and associated complaints. The report indicates that excavation of all the organic soil or peat for 30 feet around the wells occurred and replacement with clean, bank-run sand and gravel corrected the coliform contamination. The order to chlorinate was not rescinded, so the wells were not used except in emergencies because of the taste and odor problems. In later years, the increased water demands of the City made it necessary to use wells G and H despite the complaints.

Dufresne-Henry conducted air injection tests using an injection well installed about 30 feet from the gravel packed wells terminating at the same depth, and a series of observation wells. The injections were conducted to improve water quality through aeration. Results indicated that the introduction of air improved the quality of the groundwater, with manganese content dropping from 0.80 mg/L to 0.02 mg/L at a depth of 20 feet, for example based on data from monitoring wells installed near wells G and H as part of the aeration study. Reductions in nitrates were also observed. Implementation of a full-size air injection/diffuser system led to no significant change in water quality at Well G. Nonetheless, the amount of chlorine required to maintain a chlorine residual was noticeably reduced, which Dufresne-Henry identified as a satisfactory result.

Dufresne-Henry noted higher concentrations of iron in Well H compared to Well G, although concentrations of manganese were comparable. Dufresne-Henry suggested the location of Well H in the swamp as a logical explanation. Dufresne-Henry also alluded to the potential presence of organic contamination in wells G and H including “natural substances, insecticides, herbicides, and other agricultural chemicals...” (The high iron concentrations reported by Dufresne-Henry in the summer of 1977 were accompanied by a sulfide smell, which would accompany reducing groundwater.)

Dufresne-Henry determined that the odor complaints were attributable to the action of chlorine on organic matter residing in the distribution system. Samples of raw water taken directly from municipal wells G and H had no objectionable odors. Dufresne-Henry noted that manganese is present in the groundwater at wells G and H at a concentration of 0.6 milligrams per liter (mg/L) and advised that the Massachusetts Department of Environmental Quality Engineering (now the Department of Environmental Protection) would like to have the manganese removed “to a more acceptable level of 0.05 mg/L.” Dufresne-Henry recommended the construction of a treatment plant (\$1.5 million estimated) and the installation of a gravel packed well between G and H “so that the full capacity of the groundwater resource which exists in the Aberjona River Valley can be utilized.”

3.2.2 *Arsenic Species as an Indicator of Redox Conditions in Groundwater* ***(Cherry et al, 1979)***

Cherry et al (1979) identified the potential for arsenic species distributions in natural water to serve as a proxy redox indicator, particularly in groundwater, which is characterized by long equilibration time and a favorable pH-redox domain. Cherry et al cite the redox transformations

of arsenic species as sufficiently slow to enable sample collection and laboratory analysis before excessive change in species distribution, but sufficiently rapid for arsenic species to adjust to the dominant redox condition of the water. Arsenite/arsenate ratios change over a period of days, rather than seconds or minutes. Cherry et al mentioned the potential interferences of sulfide sequestration on arsenic concentrations, but noted that ferrous iron (Fe[II]) activity would limit sulfide activity such that arsenic sulfide would not reach saturation. However, Cherry et al stated that in some situations, dissolved arsenic concentrations would be limited by adsorption.

3.2.3 Adsorption of Arsenite and Arsenate on Amorphous Iron Hydroxide (Pierce and Moore, 1982)

Pierce and Moore (1982) illustrate one of the main factors controlling arsenic in natural aquatic systems: adsorption on sediments. The dependency of arsenic adsorption on concentration, amount of sorptive oxides, pH, and the oxidation state of arsenic is addressed. Amorphous iron hydroxide, which has a universal presence in clays, soils and sediments, possesses significant adsorptive properties and plays a significant role in arsenic adsorption. The extremely high adsorptive capacity of amorphous iron hydroxide for arsenic is explained by a loose, highly hydrated structure that allows ions to readily diffuse through the structure, and therefore does not limit adsorption to external surfaces as with more crystalline solids. Pierce and Moore also explain that the adsorption of arsenic to amorphous iron hydroxide indicates a specific adsorption or formation of chemical bonds between arsenic species and the adsorbent, rather than the very rapid electrostatic attraction; thus, identifying a kinetic element in the adsorption process.

3.2.4 Fixation, Transformation, and Mobilization of Arsenic in Sediments (Brannon and Patrick, 1987)

Brannon and Patrick (1987) studied the fixation, speciation, and mobilization of arsenic in sediment during sediment/water interactions, with an emphasis on the following:

- Transformation and fixation of As(V) in anaerobic sediment;
- Long-term (6-month) release from sediment of naturally occurring and added arsenic; and
- Sediment properties affecting mobilization of As(V), As(III), and organic arsenic.

Brannon and Patrick reported results from studies with dredged sediments from a variety of fresh and salt water harbor environments that included ten locations in Indiana, Wisconsin, Michigan, California, Connecticut, Texas and Washington State. Brannon and Patrick found that arsenic could be mobilized over both the short and long term. Short-term releases were found to be related to arsenic concentrations in the interstitial water and exchangeable phases of the sediment. Long-term net mass releases were related to total and extractable iron, and the calcium carbonate (CaCO₃) equivalent concentration. Redox potential and pH also played a significant role in the transformation of arsenic.

Testing involved, in part, the addition of arsenic to sediments. Under oxidizing conditions, a maximum of 17.1-percent of added arsenic remained in solution compared to a maximum of

69.5-percent under reduced conditions, with As(III) constituting all the soluble arsenic under anaerobic conditions at pH 5 to 6.5. Under conditions where anaerobic sediment is in intimate contact with anaerobic water, releases of As(III) to the water should occur. When anaerobic sediments (freshwater and saline environments) were subject to short term mixing, such as during hydraulic dredging operations or propeller wash from passing ships, As(III) was the major arsenic species released. However, leaching experiments indicated that releases of arsenic will persist for at least six months (the longest period tested) under agitated, aerobic conditions.

The release of arsenic species was phased during aerobic leaching conditions, with As(III) predominating initially (first 1 to 2 months) and As(V) becoming the predominant species as leaching progressed. Simulation of situations where anaerobic and aerobic conditions may alternate, such as in areas subject to periodic inundation, showed almost the exclusive release of As(V). These results indicated that releases of the more toxic As(III) could be curtailed by alternating anaerobic and aerobic conditions. [This alternation of aerobic and anaerobic conditions is consistent with conditions in the wetland and upper peat layers at Wells G and H, where the degree of wetland peat layer inundation varies with meteorologic conditions, river stage, and the influence of water supply well withdrawals.]

Brannon and Patrick also noted that iron oxides and hydroxides strongly adsorb arsenic compounds, and that iron and aluminum have been demonstrated to be major sinks of added arsenic in aerobic soils. Increases in arsenic concentrations in the moderately reducible phase extractant were paralleled by increased iron concentrations in the moderately reducible phase extractant. This indicated that arsenic shifts to more immobile phases during aerobic leaching, thus considerably reducing the mobility of sediment-borne arsenic and the potential for dissolved phase arsenic release.

3.2.5 Partitioning of Arsenic and Metals in Reducing Sulfidic Sediments (Moore et al, 1988)

Moore et al (1988) explored the partitioning behavior of arsenic, copper, and zinc in sediment of a Montana reservoir, which was identified as a source of contamination in an adjacent alluvial aquifer. In this study, arsenic transferred to groundwater by formation of diagenetic sulfides in sediment. This study identified a redox interface in the sediments that controlled the partitioning of arsenic and other metals. Pore water concentrations of arsenic, copper and zinc were controlled by the solubility of iron and manganese oxyhydroxides in the oxidized zone and by metal sulfides in the reduced zone. Metal concentrations (Fe, Mn, As) increased as oxyhydroxides dissolved in the reduced zone. Diagenetic sulfides scavenged these elements resulting in enrichment relative to surface (shallower) sediments. Consumption of dissolved oxygen and precipitation of sulfide altered oxidation states of trace elements in pore water and sediment and, for example, reduced As (V) to the more toxic As(III). Chemolithic bacteria mediated the reactions. Moore et al noted that 1-percent carbon was sufficient to drive bacterial reduction and increase the rate of sulfide reactions. In this scenario, trace elements stored by formation of authigenic sulfides were a potential source of secondary contamination if sulfides were moved into oxidizing environments where they are unstable. Chemolithotrophic bacteria assisted in this sulfide oxidation process.

3.2.6 Arsenic and Selenium Chemistry as Affected by Sediment Redox Potential and pH (Masscheleyn et al, 1991a)

Masscheleyn et al (1991a) identified redox conditions that can limit or enhance the translocation and movement of arsenic and selenium in the sediment-water environment. Redox potential and pH affect both speciation and solubility of arsenic and selenium in reservoir sediments. Under highly oxidized conditions, arsenic solubility was low and most arsenic in solution was arsenate, As(V). Upon reduction, As(III) became the major arsenic species in solution and arsenic solubility increased substantially. The data suggest that the increase in soluble arsenic was due to reductive dissolution of iron oxyhydroxides, which strongly adsorb As(V). Arsenic solubility increased 25 times upon reduction from a redox potential of 500 millivolts (mV) to -200 mV. More than half the arsenic present in sediment was soluble at -200 mV. Masscheleyn et al noted that precipitation/dissolution reactions of arsenic minerals were not the controlling factors in arsenic solubility; rather, arsenic chemistry is controlled by adsorption-desorption mechanisms. Also, total water-soluble arsenic and iron were highly correlated, while there was no correlation between aluminum and manganese, suggesting that aluminum and manganese oxides are less important in controlling arsenic solubility.

3.2.7 Effect of Redox Potential and pH on Arsenic Speciation and Solubility in a Contaminated Soil (Masscheleyn et al, 1991b)

Masscheleyn et al (1991b) investigated the control of redox and pH on arsenic speciation and solubility and explored arsenic behavior over a range of redox and pH conditions. For example, under moderately reducing conditions (0 to -100 mV), arsenic solubility was controlled by the dissolution of iron oxyhydroxides. Arsenic can be co-precipitated as As(V) with iron oxyhydroxides and subsequently released upon their solubilization under reducing conditions. Masscheleyn et al observed slow kinetics for the As(V) -As(III) transformations, which can explain, for example, why thermodynamically unstable As(V) species can be observed under reducing conditions. Nonetheless, Masscheleyn et al advised that high redox and non-alkaline conditions minimize solubility and mobilization of arsenic.

3.2.8 The Distribution and Depositional History of Metals in Surface Sediments of the Aberjona River Watershed (Knox, 1991)

Knox (1991) studied the transport of primarily arsenic and associated water quality parameters in the Mystic Lake Upper Forebay in Winchester, Massachusetts during the period from June 1995 to July 1996. Sediment and surface water were sampled for a variety of chemical constituents. Through radiological dating, the sedimentation rate was determined to be approximately 0.42 to 0.6 centimeters per year in the upper forebay. Sampling was conducted during average base flow conditions and during storm events. Knox noted that during the summer suspended solids and arsenic levels were higher in the outflow from the forebay in comparison to the inflow. This led to the theory that the higher suspended solids may be due to plant matter and the high arsenic concentrations may be due to arsenic attached to plant matter, arsenic released by plant metabolism activity, or due to groundwater influx containing elevated arsenic levels. In the winter there was a reduction in suspended solids and arsenic in the discharge. During storm events more arsenic was attached to the larger suspended sediments (>8 microns) than present in

smaller particulates (between 0.45 and 8 microns). During high flow events there were two distinct arsenic loading pulses. The first pulse was due to drainage from the more southerly Winchester sub-basin and the second pulse was from the upper portions of the basin Woburn-Central and Woburn-North areas and, in particular, the HBHA. Knox referred to the first pulse as the quick storm response and the later pulse as the slow storm response, and calculated that the first and second pulses were responsible for discharging about 5.7 pounds and 7.5 pounds of total arsenic to the Upper Forebay during a January storm. The second pulse had more dissolved arsenic and the particulates were smaller in comparison to the first pulse. Based on a range of sediment arsenic data, the rate of arsenic burial in forebay sediments was between 25 and 93 pounds per year. Over the course of a year, the amount of dissolved and particulate arsenic entering the Forebay appeared to be about equal. The amount of particulates accumulating in the Forebay was about 65-percent. The amount of arsenic entering the Forebay on an annual basis was approximately 34 to 153 pounds per year. Solo-Gabriele (1995) showed yearly, average, arsenic releases to the Aberjona watershed of 600 pounds per year for the period from 1900 to 1993. The highest fluxes were observed in the 1920s, 1950s and 1960s.

3.2.9 Arsenic in the Aberjona Watershed (Aurilio, 1992)

Aurilio (1992) examined the source, distribution, and speciation of arsenic in the Aberjona River watershed, with a focus on the HBHA and the Upper and Lower Mystic Lakes. Aurilio estimated that 300 metric tons of arsenic were present in the contaminated areas of the watershed, and used manufacturing data relevant for the period of sulfuric acid, arsenical pesticide, leather, glue, and gelatin manufacturing in the Aberjona River valley to conclude that 200 to 700 metric tons of arsenic may have been released as a result of these activities. Approximately 10 metric tons migrated over 10 kilometers from the site of origin. The largest quantity of arsenical waste is attributable to sulfuric acid manufacturing, followed by arsenical pesticide manufacturing, which took place at the Industri-Plex Site. Pyrite ores used in acid manufacturing contained arsenic and other metals as impurities.

Aurilio described the complex geochemistry of arsenic, noting that the two most prevalent arsenic species found within the pH range of natural waters are arsenite (As[III]), which is thermodynamically favored in acidic and anoxic waters, and arsenate (As[V]), which is prevalent in oxic waters. Reduction and methylation of arsenic (arsenite, dimethylarsenate, monomethylarsenate) may lead to increased mobilization of arsenic, since these forms are much less particle reactive than arsenate. Oxidation of arsenic to arsenate (As[V]) and adsorptive scavenging by iron oxides exerts significant control on arsenic solubility under oxidizing conditions.

Aurilio reviewed the complex factors controlling the fate of arsenic in the HBHA and the Upper and Lower Mystic Lakes, noting that arsenic speciation does not always correspond to purely thermodynamic considerations. Aurilio encountered and described diverse chemical, physical, and biological controls on the mobility of arsenic.

3.2.10 Characterization of the Physical and Engineering Properties of the Aberjona Wetland Sediments (Bialon, 1995)

Bialon (1995) conducted a study of the controlling mechanisms of the hydraulic properties of the Aberjona wetland sediment as a Masters Thesis at the Massachusetts Institute of Technology. The study was based on the results of 35 constant rate of strain consolidation (CRSC) tests, 2 cylindrical specimen triaxial-cell permeameter tests, and 4 cubical specimen triaxial-cell permeameter tests. Specimens were sampled from three locations within the Aberjona wetland by a manually operated fixed-piston sampler. The work was conducted in the wetland to the northwest and west of Well H near the Aberjona River.

Bialon observed that the wetland sediment was divided into five distinctive layers based primarily on a visual classification: live root mat, a typha peat layer, sedge peat layer, read woody peat layer, and the diatomaceous earth layer. A range of properties was presented for each of the layers.

The range of compressibility ratios was very comparable between the wetland layers. Bialon found that the Compressibility Ratio (CR) and the Recompression Ratio (RR) of the peat layer ranged from (0.275 to 0.980) and (0.013 to 0.096), respectively. Bialon noted that the compressibility properties were much lower in the diatomaceous earth, with the values of CR and RR of (0.224 to 0.412) and (0.002 and 0.016), respectively.

Bialon concluded that there appeared to be no significant difference in hydraulic conductivity between the wetland layers and that the range of in-situ hydraulic conductivity throughout the wetland layers is between $2.0\text{E-}6$ centimeters per second (cm/sec) and $6.0\text{E-}4$ cm/sec. Study results suggested that the in situ hydraulic conductivity of the wetland layers was not related to void ratio. Bialon noted that organically bound pore water influenced the variability in hydraulic conductivity and found that the amount of bound pore water increased with increases in the organic content of sediments.

3.2.11 Relation of Arsenic, Iron, and Manganese in Groundwater to Aquifer Type, Bedrock Lithochemistry, and Land Use in the New England Coastal Basins (Ayotte et al, 1999)

The Ayotte et al (1999) USGS study examined arsenic, iron and manganese data from public water supply wells in the New England Coastal Basin and determined that the concentration of arsenic in bedrock groundwater varies with bedrock lithology, but also with land use (i.e., higher in agricultural areas due to the use of arsenical pesticides). Ayotte et al found the lithochemical relation to be statistically stronger than the land use relation, but also acknowledged the bias in their data set to public water supply wells. Non-potable groundwaters (those with high dissolved solids or with water that does not meet regulatory standards) could differ with respect to spatial and chemical association. Nonetheless, Ayotte et al (1979) found that arsenic was detected in 25.5-percent of public supply wells in bedrock and in 7.6-percent of stratified drift aquifers, noting that the significant difference may result from the more evolved chemistry of bedrock aquifers (longer residence time). Groundwater residence time in stratified drift aquifers was relatively short and more readily subject to recharge from the surface. Ayotte

et al found that high iron and manganese concentrations were not always a good indicator of high arsenic concentrations in the bedrock supply wells, suggesting that simple dissolution of iron sulfides and hydroxides did not account for arsenic concentrations in bedrock groundwater.

3.2.12 Arsenic in Groundwater of the Willamette Basin, Oregon – Geochemistry of Arsenic (Hinkle and Polette, 1999)

Hinkle and Polette (1999) conducted a study as part of the USGS in 1996 designed to increase the current understanding of the groundwater resource, and to better characterize the distribution of naturally occurring poor quality groundwater in the Willamette Basin, Oregon, in response to increasing demands on groundwater resources. Arsenic concentrations exceeding the then current EPA MCL of 50 ug/L are widespread in groundwater in the Willamette Basin.

Hinkle and Polette described factors controlling the distribution of arsenic in groundwater such as arsenic sources and processes controlling arsenic mobility. Processes that have been shown to control arsenic mobility in natural systems were discussed. Arsenic speciation data collected as part of this project, along with some historical speciation data, were presented. Finally, geochemical data (including the speciation data) and information from existing interpretive reports were used to construct preliminary hypotheses regarding possible geochemical controls over mobilization of arsenic in the Willamette Basin.

Hinkle and Polette described two categories of processes that largely control arsenic mobility in aquifers: (1) adsorption and desorption reactions and (2) solid-phase precipitation and dissolution reactions. Arsenic adsorption and desorption reactions are influenced by changes in pH, occurrence of redox (reduction/oxidation) reactions, presence of competing anions, and solid-phase structural changes at the atomic level. Solid-phase precipitation and dissolution reactions are controlled by solution chemistry, including pH, redox state, and chemical composition. Existing data, including the speciation data, and published interpretations were used to establish preliminary hypotheses for the evolution of high-arsenic groundwater in the Willamette Basin.

3.2.13 Geochemical and Paleohydrological Controls on Pollution of Groundwater by Arsenic (Ravenscroft et al, 2001)

Ravenscroft et al (2001) examined the occurrence of arsenic in groundwater in the Bengal Basin of Bangladesh and West Bengal (India), where they identify the reduction of iron oxyhydroxide (FeOOH) and the release of its sorbed arsenic load to solution as an important mechanism by which groundwater world wide becomes polluted with arsenic. Ravenscroft et al assigned an important role to organic matter, in particular peaty sediments, in generating anoxic conditions in groundwater, identifying peat as the redox driver for reduction of FeOOH. The distribution of pollution by arsenic reflects the distribution of buried peat deposits. Ravenscroft et al postulated that the pollutant arsenic in groundwater occurs where microbial reduction of FeOOH has released sorbed arsenic and also reduces the arsenic. Organic matter is required to drive this process. They further postulated that the distribution of organic matter in the aquifer sediments was the main control on the distribution of arsenic pollution in their study area. They also found that poor correlations of arsenic with iron may arise because of re-sorption (partial, since arsenite

adsorbs less well than does arsenate) of arsenic on to freshly exposed FeOOH surfaces exposed by dissolution or because the arsenic/FeOOH ratio may vary from place to place in response to differing amounts of FeOOH and mineral surface, and differing mineral abundances.

3.2.14 Validation of Arsenic Sequential Extraction Method for Evaluating Mobility in Sediments (Keon et al, 2001)

Arsenic mobility and transport in the environment are strongly influenced by the association of arsenic with solid phases in soil and sediment. Keon et al (2001) tested a sequential extraction procedure intended to differentiate the following pools of solid phase arsenic: loosely and strongly adsorbed arsenic; arsenic coprecipitated with metal oxides or amorphous monosulfides; arsenic coprecipitated with crystalline iron (oxyhydr)-oxides; arsenic oxides; arsenic coprecipitated with pyrite; and arsenic sulfides. Additions of arsenic-bearing phases to wetland and riverbed sediment subsamples were quantitatively recovered by a sequential extraction procedure.

Wet sediment subsamples from both highly contaminated wetland peat and less arsenic-rich sandy riverbed sediment from Wells G and H were used to test the extraction procedure. The proportional distribution of arsenic among extractant pools was consistent for subsamples of the wetland and for subsamples of the riverbed sediments.

3.2.15 Use of Hydrochloric Acid for Determining Solid-Phase Arsenic Partitioning in Sulfidic Sediments (Wilkin and Ford, 2002)

Wilkin and Ford (2002) examined the use of room-temperature hydrochloric acid and salt solutions of magnesium chloride, sodium carbonate, and sodium sulfide for the removal of arsenic from synthetic iron monosulfides and contaminated sediments containing acid-soluble arsenic reacts with hydrogen sulfide released from Acid Volatile Sulfide (AVS) phases and precipitates at low pH as disordered orpiment or alacranite.

Arsenic sulfide precipitation is consistent with geochemical modeling in that conditions during acid extraction are predicted to be oversaturated with respect to orpiment, realgar, or both. Wilkin and Ford found that acid extraction of arsenic from sulfide-bearing sediments will give biased results that overestimate the stability or underestimate the bioavailability of sediment-bound arsenic. Alkaline solutions of sodium sulfide and sodium carbonate are efficient in removing arsenic from arsenic sulfides and mixed iron. Arsenic sulfides are more efficiently removed because of the high solubility of arsenic at alkaline pH, the formation of stable arsenic complexes with sulfide or carbonate, or both.

3.2.16 Speciation of Arsenic in Sulfidic Waters (Wilkin et al, 2003)

Wilkin et al (2003) examined the change in As(III) speciation from arsenite to a distribution of arsenic-sulfide complexes (thioarsenic species) with increase sulfide concentrations in aqueous solution. Conversion from neutral arsenate species $[\text{As}(\text{OH}_3)^0]$ to anionic thioarsenite species may regulate the transport and fate of arsenic in sulfate-reducing environments by governing sorption and mineral precipitation reactions. However, concentrations of dissolved sulfide are

typically low when reactive iron is abundant. Dissolved sulfide concentrations are able to increase only after supply of reactive iron is exhausted via reductive dissolution processes and subsequent iron monosulfide precipitation. The presence of reactive iron will generally preclude the formation of thioarsenic species in sulfate-reducing systems. Environments where the abundance of reactive iron is limited will favor sulfide accumulation and thioarsenite formation.

3.2.17 The Impact of Groundwater-Surface Water Interactions on Contaminant Transport at Contaminated Sites – Draft (Ford, 2004a)

Ford (2004a) studied the impact of groundwater-surface water interactions on contaminant transport in the HBHA at the Industri-Plex Superfund Site. Ford's site monitoring strategy included the installation of tubing wells, diffusion samplers, and depth-resolved sampling within the pond to measure water chemistry within the HBHA and sampling of upgradient groundwater wells.

Ford determined that in addition to the arsenic that was flowing into the pond through groundwater discharge that "a key component that contributes to the arsenic mass balance within the (pond) water column is the internal recycling of arsenic originally derived from ground-water discharge. The internal recycling process is due to the coupling of iron oxidation-reduction processes that, in part, control the distribution of arsenic between water and solids within the water column." Ford found that arsenic enters the pond through groundwater and upon entering the pond some arsenic partitions to sediment and some contributes directly to surface water contamination. However, the amount that partitions to the sediments can reenter the water column under reducing conditions. Ford observed that this occurred following a large storm event, and postulated that arsenic contaminated sediment from the aerobic zone had been re-deposited within the pool and subjected to reducing conditions. In this particular case, the concentration of arsenic in pond water was approximately equally associated with groundwater flux and sediment releases. Ford noted that the removal of the contaminated groundwater source might not result in full cleanup of the HBHA water.

Ford (2004a) also conducted a literature search and echoed the following themes reported by the collective authors:

- Hydrologic transients imposed by long-term (seasonal) and short-term (storm events) flow variations impact the types and intensity of chemical reactions that influence contaminant chemical speciation and transport;
- Microbial degradation of natural and anthropogenic sources of organic matter and the availability of terminal electron acceptors influences the distribution of oxidation/reduction (redox) zones within the groundwater/surface water (GW/SW) transition zone;
- Microbial processes and contaminant transport are dependent on the chemical characteristics of both the liquid and solid phase across the GW/SW transition zone; and

- Chemical gradients of major and trace elements/contaminants are often greatest in the vicinity of the GW/SW transition zone.

3.2.18 Source and Behavior of Arsenic in Natural Waters (*Smedley and Kinniburgh, 2004*)

Smedley and Kinniburgh (2004) prepared a comprehensive review of arsenic sources and behavior in natural waters, and address topics such as mineral-water interactions, groundwater environments showing enhanced arsenic concentrations, and common features of groundwater arsenic problem areas. The discussion of arsenic geochemistry is current and provides and identifies outstanding questions and further needs.

4.0 SITE PHYSICAL CHARACTERIZATION

The Wells G and H Superfund Site is a 330-acre site situated in east Woburn, Massachusetts. The site is bounded by Route 128/Interstate 95 to the north, Interstate 93 to the east, Boston & Maine railroad tracks to the west, and Salem and Cedar Streets to the south (see Figure 1). Wells G and H are two municipal water supply wells located in the Aberjona River Valley that supplemented the City of Woburn's water supply in the 1960s and 1970s (EPA, 1989).

4.1 Contamination Synopsis

4.1.1 Operable Units

The OU-1 Source Area properties consist of the W.R. Grace & Company (Grace), UniFirst Corporation (UniFirst), New England Plastics (NEP), Wildwood Conservation Corporation (Wildwood), and Olympia Nominee Trust (Olympia) properties, the locations of which are depicted on Figure 1 (EPA, 2004).

The Central Area (OU-2) consists of all groundwater and land within the area defined as the Wells G and H Superfund Site, excluding the areas defined for Source Area (OU-1) properties and the Aberjona River Study (OU-3).

The groundwater aquifer underlying the Site is not currently used as a municipal drinking water source. The objectives listed in the Site ROD include restoring the aquifer to drinking water standards. Public opinion has been opposed to utilizing Wells G and H for water supply. However, the City of Woburn has expressed interest in having the source available for future use (MADEP, 2004). The MADEP's Groundwater Use and Value Determination assigned a "medium" use and value for the Site aquifer, based on a balanced consideration of several factors, and contemplates future use of the aquifer for domestic and industrial purposes (EPA, 2004).

The portion of the Central Area (OU-2) known as the Southwest Properties includes the Aberjona Auto Parts, Whitney Barrel, and Murphy Waste Oil properties. Aberjona Auto Parts began operations in the mid-1950s for the sale and reconditioning of used and wrecked automobiles, and was also a gasoline service station. Current uses of the Aberjona Autoparts property include the remnants of the now closed auto salvage operation, vehicle repair, landscaping, and a detached residence. The Whitney Barrel Company located on Salem Street commenced operations in 1949, and reconditioned drums, boilers, tanks and machinery. Current uses of the Whitney Barrel property are mixed and include landscaping firms, automotive glass repair, and miscellaneous storage. The Murphy Waste Oil property is a Resource Conservation and Recovery Act (RCRA)-permitted Treatment, Storage and Disposal Facility (TSDF) operated by Clean Harbors, Inc. (EPA, 2004).

The Aberjona River Study (OU-3) area consists of the Aberjona River and its tributaries, sediments, and associated 38-acre wetland area that lie within the 330-acres of the Site. The Aberjona River begins in Reading, Massachusetts, and flows through the Industri-Plex Superfund Site to the north of Route 128 before flowing through the Site, and eventually reaches

the Mystic Lakes in Winchester. The investigation of the Aberjona River was recently merged with the Industri-Plex Superfund Site and the Aberjona River study area now extends to the Mystic Lakes in Winchester.

4.1.2 History of Contamination

On May 4, 1979, 184 55-gallon drums containing polyurethane and toluene diisocyanate were found on Mishawum Road on a vacant lot owned by the Massachusetts Bay Transportation Authority (MBTA). The drums were removed during negotiations with the Massachusetts Department of Environmental Quality Engineering (DEQE) (now the MADEP). The drum discovery prompted the DEQE to sample the nearest downgradient public water supply, Wells G and H. Several chlorinated VOCs were detected in water from Wells G and H at concentrations ranging from 1 to 400 ug/L. The City of Woburn was forced to use Metropolitan District Commission (MDC) water to supplement its public water supply when Wells G and H were shut down on May 21, 1979. The MDC (now the Massachusetts Water Resources Authority or MWRA) continues to supplement the City of Woburn's water supply (EPA, 2004).

EPA and various property owners have conducted numerous studies to determine the nature and extent of contamination at the Site. The following five facilities have been identified as sources of contamination – Grace, UniFirst, NEP, Wildwood, and Olympia. Wells G and H Superfund Site was listed as a Superfund Site on the National Priorities List (NPL) on December 21, 1982 (EPA, 2004).

4.1.3 Contaminant Summary

The following summarizes the contaminants detected at the Site as identified in the ROD (EPA, 2004).

Groundwater. Chlorinated VOCs are the primary groundwater contaminants. Groundwater contamination has been found in overburden and bedrock aquifers at the Grace, UniFirst, Wildwood and NEP properties as well as the Central Area (OU-2) of the Site. Groundwater contamination has been found in the overburden aquifer at the Olympia FDDA.

The Grace contamination consists primarily of chlorinated solvents characterized by a high percentage of trichloroethene (TCE) and 1,2-dichloroethene (DCE). Other contaminants include PCE and vinyl chloride. The UniFirst contamination is predominantly PCE. Secondary constituents are 1,1,1-trichloroethane (1,1,1-TCA), and smaller amounts of TCE and 1,2-DCE. The Wildwood contamination consists primarily of TCE detected at a number of wells, with 1,1,1-TCA, 1,2-dichloroethene (DCE), and PCE detected at a few locations. At Olympia, TCE and xylene were detected in the overburden. At NEP, PCE, TCE, 1,1,1-TCA and 1,2-DCE were found in bedrock and overburden wells.

Soil. Chlorinated VOCs are the primary contaminants in soil and were found at various levels on the Wildwood, Olympia, Grace, NEP and UniFirst properties. Some chlorinated VOC soil contamination was also found in a wetland area at Wildwood.

Other soil contaminants include PCBs, chlordane, phthalates, and polyaromatic hydrocarbons (PAHs), which were found dispersed throughout the Wildwood property. PAHs were found in one location at Olympia. Phthalates were found in a small area at NEP. Assorted debris and sludge contaminated with lead, VOCs, PAHs, and pesticides were also found at Wildwood.

Sediment/River. Aberjona River and wetland sediments were contaminated with PAHs, PCBs, pesticides, and metals such as arsenic, copper, mercury, zinc, and chromium. Surface water samples revealed low levels of chlorinated VOCs. Metals and phthalates were also noted in surface water.

Air. Air monitoring, conducted during all site investigations, did not reveal any VOC readings above background at the breathing zone.

Potential health risks identified at the Site include ingestion of contaminated groundwater, inhalation of volatiles while showering, and dermal contact or incidental ingestion of surface soils (EPA, 1989). Arsenic in sediment was identified as contributing to risk above a level of concern for recreational site use. For ecological receptors, the evaluation indicated potential risk to aquatic life due to metals and phthalates in surface water. Potential risk to invertebrates and mammals were identified due to metals, pesticides, PAHs, and PCBs in sediments (EPA, 2004).

4.2 Aberjona River

The Aberjona River headwaters are in west-central Reading, Massachusetts. The River flows southwest from Reading through the City of Woburn and the Town of Winchester before entering the Mystic Lakes system. The Aberjona River passes through a mix of parkland, residential, urban, and light industrial areas, with the industrial areas found largely in the City of Woburn. Consequently, the river and associated water bodies and wetlands are affected by a number of potential factors, including neglect, indiscriminant disposal of debris, local and upstream runoff, including non-point and point source discharges. Development, loss of flood storage, culverting, and channelization has also impacted the river (M&E, 2003).

In 1985, the USGS conducted a 30-day aquifer pump test to determine the zone of contribution to Wells G and H (USGS, 1987). The study concluded that a hydraulic connection between the aquifer and the river exists under pumping conditions. Also in 1985, EPA determined the extent and type of wetlands that exist at the study area (PRC, 1986)

Massachusetts regulations classify the Aberjona River as Class B surface water that supports fish populations. Fish sampling confirmed the presence of warm water species throughout the study area. Class B waters are defined by the MADEP as “a habitat for fish, other aquatic life and wildlife, and for primary and secondary recreation. Where designated they shall be suitable as a source of public water supply with appropriate treatment. They shall be suitable for irrigation and other agricultural uses and for compatible industrial cooling and process uses. These waters shall have consistently good aesthetic value.” “Primary contact recreation” represents “any recreation or other water use in which there is prolonged and intimate contact with water and a significant risk of ingestion. These include, but are not limited to, wading, swimming, diving, surfing and water skiing.” “Secondary contact recreation” represents “any recreation or other

water use in which contact with water is either incidental or accidental. These include but are not limited to fishing, boating, and limited contact incidental to shoreline activities.” (MADEP, 1996 and 1997)

In slow-moving sections, the river meanders through vegetated wetlands (HBHA north of Route 128 and the Wells G and H 38 acre wetland in Woburn). In some of the more urban areas (Route 128 area and in downtown Winchester), the river is culverted or artificially channeled (M&E, 2003).

The population has grown in recent years in the City of Woburn from 35,835 in 1990 to 37,528 in 2000 (M&E, 2004) and from 21,221 in 1990 to 21,344 in 2000 in the Town of Winchester (M&E, 2004) with the expansion of light and technology-related industries along the major interstate highways (Routes 93 and 95/128). The overall increase in population and development has placed an additional demand on the use of open space areas, particularly on the parklands bordering the river in the area of the Woburn-Winchester town line and along the lower reaches of the Aberjona River as it enters the Upper Mystic Lake. Future land use is not expected to change significantly (M&E, 2004).

Wildlife habitat associated with the river and water bodies within the study area is generally restricted to a relatively narrow corridor. The width of this corridor varies from approximately 20 feet to 0.3 miles. At several locations, development encroaches to the waters edge. Habitats along the river include emergent, scrub/shrub and forested wetlands, fragmented upland forests, sub-mature woodlots, grassy meadows, and maintained parkland. The study area is in an urban watershed. In areas not directly impacted by human activities through alterations of the riverbank or channelization, the river habitat is indirectly influenced by stormwater run-off or proximity to human activity, which can affect habitat quality for some species (M&E, 2004).

5.0 RELEVANT CHEMICAL BEHAVIOR IN THE ENVIRONMENT

The primary contaminants of interest for this analysis are arsenic, copper, chromium, mercury and lead, which were identified as risk drivers in the September 2004 *Baseline Human Health and Ecological Risk Assessment Report* prepared for EPA (M&E,2004). PAHs, specifically benzo(a)pyrene, were identified as minor contributors to risk. PAHs will not be addressed in this analysis since the fate and transport behavior of PAHs is characterized by limited migration potential in groundwater, and very limited phase transfer from sediments to groundwater. The presence of chlorinated VOCs in the aquifer at concentrations that drive risk is widely documented at Wells G and H. Chlorinated VOCs in groundwater were not evaluated as part of this analysis.

5.1 Metals Fate and Transport

The potential release and migration of metals in the subsurface environment is a complicated process. The mobility of metals depends on factors such as the overall groundwater composition, pH, metal complex formation, valence state of the metal, and cation-anion exchange capacity. Changes in the oxidation-reduction potential (ORP) in soil or groundwater can affect the chemical species present. Metals occur in the environment as free ions or as complexed species. Complexing generally increases the solubility and mobility of metals in groundwater. Additionally, the type of complex a metal forms depends on whether the species is hard (strongly held electron field) or soft (deformable electron field). Hard species form stronger bonds than soft species. In general, complexation reduces metal toxicity (Campbell, 1995; DiToro et al, 1990).

The distribution between soil and water for metals is much more difficult to estimate than for organic compounds. Since the sorption of metals depends on pH, the metal concentration, the species present, and the type of complex formation, a single distribution coefficient or isotherm equation cannot be used to predict metal adsorption. Literature aqueous partitioning coefficient (K_d) values can vary by more than two orders of magnitude (ERG, 2003). Generally, metal adsorption increases with pH. Inorganics most often sorb to clay minerals, organic matter, and iron and manganese oxyhydroxides. The surface charge of organic matter and oxyhydroxides is strongly pH-dependent, becoming more negative as pH increases and more positive as pH decreases. Metals may be sorbed on the surface of the soil or fixed to the interior of the soil, where they are unavailable for release to water. After available sorption sites are filled, most metals are incorporated into the structures of major mineral precipitates, as coprecipitates. At very high concentrations, they may be precipitated into pure metal phases.

The solubility of metals is also dependent on several factors. The solubility of cations decreases as pH increases. For a few cations (e.g., Zn⁺², Al⁺³, and Fe⁺²), metal solubility increases again at alkaline pH values. The solubility of metals may decrease depending on the complex formation. Some cations may complex with oxygen and hydroxide, forming insoluble oxyhydroxides, or may complex with phosphate, sulfate, and carbonate to form insoluble mineral precipitates. Metal sulfide complexes, which form in reducing environments, are extremely insoluble, and their formation tends to reduce the total metals concentrations (EPA, 1979). However, arsenic is the exception to the insolubility of metal sulfides at circumneutral pH (Wilken et al, 2003).

Metals may be removed from the water phase through mechanisms such as precipitation and irreversible sorption (EPA, 1979).

In the water phase, the total metal concentration includes the dissolved metal concentration and the suspended metal concentration, which is sorbed to colloidal particles. As a result, the groundwater data may reflect metals concentrations that are associated with a significant percentage of colloidal material. Although some groundwater samples from the Wells G and H Site were filtered with a 0.45-micron filter, studies indicate that the most mobile particles were in the range of 0.1 to 0.55 micron and contributed as much as 40 percent of the total mobile metals (EPA, 1979). Therefore, elevated metals concentrations in groundwater may be due to the suspended load and not just to the dissolved percentage.

The fate and transport properties of metals identified as being potential risk drivers in the Aberjona River Study (M&E, 2003) are discussed in more detail below.

5.1.1 Iron and Manganese

Iron (Fe) and manganese (Mn) geochemistry are important to the understanding of Central Area aquifer geochemistry because of their ubiquitous presence, controlling influence on the solubility of arsenic, and because their presence is indicative of the overall redox status of the aquifer. The following summary of iron and manganese fate and transport properties is important to the interpretation of the available iron and manganese data.

Iron and manganese are naturally occurring elements that are ubiquitous in the environment. Manganese is a hard cation, which is often precipitated in soils to manganese minerals. Iron is a hard cation in the Fe^{+3} oxidation state and a borderline cation in the Fe^{+2} oxidation state. The transport of these elements is dependent on their species and the pH and ORP of the soil or water environment (ERG, 2003). Low pH and redox potential favors ferrous iron (Fe[II]); high pH and/or redox levels create a stable environment for ferric iron (Fe [III]) (Bodek et al, 1988).

In aqueous solution, ferric and ferrous iron hydrolyze to give various hydrolysis species, the distribution of which is a function of pH (Bodek et al, 1988). Both iron (III) oxyhydroxides and manganese oxides are relatively insoluble in oxidizing environments and are strong sorbants of other metals (ERG, 2003). In natural streams, iron occurs as particulate ferric hydroxide or as some form of organic complex. Ferrous iron is the most common form of dissolved iron in groundwater. Colloidal ferric hydroxide is typically present in surface waters (Bodek et al, 1988).

In lakes, ponds, and rivers, water near the bottom may be depleted of oxygen and attain a low oxidation-reduction potential (Eh), and may contain dissolved ferrous (reduced) iron concentrations up to several mg/L. Suspended iron sulfides that can form in the depleted oxygen/low Eh environment may be carried upward by mixing processes and become oxidized when they reach the oxygen containing waters. Subsequent hydrolysis forms amorphous ferric hydroxides, which may be colloidal and remain suspended at the boundary layer between the reduced water and the surface oxygenated water. With further coagulation, the larger particles settle into the anoxic waters, are converted to sulfides, and return to the bottom sediments

(Bodek et al, 1988). Aurilio (1992) described similar phenomena in the Lower Mystic Lake, which led to the formation of a highly turbid boundary layer between aerobic and anaerobic zones. Ford (2004b) also observed a turbidity maximum at the aerobic/anaerobic interface in the HBHA attributable, in part, to an observed maximum in particulate iron.

Iron and manganese oxyhydroxides and oxides can be used by microorganisms as electron acceptors under reducing conditions and are reduced to more soluble forms in a process known as bioreduction or reductive dissolution (ERG, 2003).

Manganese is often precipitated in soils and sediments as manganese minerals. Environmentally, the most important of these minerals are the Mn(III/IV) oxides. These phases are stable and relatively insoluble in oxidized systems above pH 6–7. The manganese oxides are often stronger sorbents of trace metals than are hydrous ferric oxides. Most of the cobalt, nickel, copper, zinc, lead, silver, and cadmium in a Pennsylvania soil were associated with manganese and iron oxides, with the manganese oxides holding the highest metal amounts relative to their abundance (ERG, 2003).

The generally observed Eh-pH regimes for iron and manganese in Central Area aquifer groundwater are illustrated in Figures 3 and 4, respectively. The Eh-pH diagrams in Figures 3 and 4 are graphical representations of aqueous oxidation-reduction (redox) equilibria for the featured inorganic species (in these cases iron and manganese, respectively). The Eh-pH diagrams graphically depict areas of predominance for in an Eh-pH coordinate system for various aqueous chemical species involved in redox, acid-base, precipitation, and complexation equilibria. Each Eh-pH diagram assumes one set of pressure and temperature conditions, and can be constructed for site-specific chemical, temperature, pressure, and concentration regimes. The Eh-pH plots provided herein are based on a water temperature of 25°C. Although groundwater is typically cooler, the 10°C± difference between the Eh-pH plots and typical groundwater temperature (i.e., 15°C) will negligibly affect the depicted equilibrium relationships. The generic Eh-pH diagrams presented herein are nonetheless useful for conceptually illustrating fundamental aqueous equilibrium relationships. For example the site groundwater Eh-pH conditions superimposed on the iron Eh-pH diagram (Figure 3) favor the presence of both soluble and insoluble iron, with a significant fraction of iron hydroxide solids [Fe(OH)_{3(s)}]. Hydrous iron oxides have high adsorptive capacity for arsenic and high specific surface area, and therefore are exceptional sinks for arsenic. Consequently, iron oxyhydroxides should exert significant control on arsenic solubility. In contrast, the site groundwater Eh-pH conditions superimposed on the manganese Eh-pH diagram (Figure 4) suggest that manganese is largely soluble even in the presence of sulfur and carbon.

Note that over 90-percent of the manganese analyses from groundwater samples collected on behalf of EPA in 2002 using low stress (low flow) purging protocols outlined by EPA Region 1 exceeded the EPA Secondary MCL (SMCL) of 50 ug/L, with a 100-percent detection rate. Iron groundwater results from these same sampling events were also significant in terms of detection rates and SMCL exceedances, with a detection rate greater than 90-percent, and with 70-percent of results in excess of the SMCL of 300 ug/L. The comparatively lower rate of SMCL exceedances may reflect the condition of the site groundwater Eh-pH regime, which suggests a significant fraction of iron oxyhydroxy solids at equilibrium. Note that both iron and manganese

are present at concentrations several orders of magnitude greater than typical arsenic groundwater concentrations. For example based on recent (2002) low flow groundwater sampling conducted by EPA in the FDDA, wetland, and Central Area (TRC, 2002a,c), the detected concentrations of iron ranged from 49.6 to 43,700 ug/L, with a median value of 1,100 ug/L. Detected concentrations of manganese ranged from and 2.8 to 9,010 ug/L, with a median value of 551 ug/L. Detected arsenic concentrations from the same sampling events ranged from 0.13 to 371 ug/L, with a median concentration of 1.9 ug/L.

Note also that iron concentrations in available soil data suggest that iron is plentiful in Central Area overburden. For example, iron in soil samples from the FDDA at the Olympia site ranged from 6,500 J mg/kg to 14,000 J mg/kg at a 100-percent rate of detection.

5.1.2 Arsenic

5.1.2.1 Arsenic Fate and Transport

Arsenic is a significant risk driver for human and ecological receptors in Aberjona River and wetland sediments near Wells G and H and in other reaches of the Aberjona River (e.g., the cranberry bog to the south of Salem Street). The presence of the arsenic in the river and wetland sediments and in the groundwater poses a potential threat to potable water development in addition to that posed by chlorinated solvents like TCE. Whether the arsenic will pose an actual threat depends in part on the geochemistry and hydrologic properties of the aquifer. The following summary of arsenic fate and transport properties is important to the interpretation of the available arsenic data and related geochemical/water quality data.

The geochemistry of arsenic is complex and has a significant impact on the mobility and toxicity of arsenic. For example, the principal forms of arsenic found in natural waters are As(V), and As(III). In general, As(V) is the oxidized form and is less toxic, less mobile, and has a greater tendency to adsorb to aquifer solids (especially iron oxyhydroxide [FeOOH]) than does the reduced form As(III). As(III) is most prevalent in reducing environments. As(III) is more toxic, more mobile, and is less likely to adsorb to aquifer solids, although adsorption does take place to a lesser degree. While the equilibrium chemistry of arsenic, especially As(V) and As(III), is well documented, the equilibrium chemistry is not simple, and equilibrium properties cannot readily be assigned to dynamic hydrogeologic or geochemical situations. Arsenic fate and transport behavior is significantly influenced by the presence of other chemical components and species, such as iron, manganese, sulfur and some organics, as well as pH, dissolved oxygen and oxidation-reduction potential. The fate and transport behavior of arsenic is further complicated by arsenic transformation reaction kinetics, which are relatively slow compared to other chemical species like iron. Arsenic redox behavior is also significantly influenced by the presence of highly organic materials, like peat.

Iron and arsenic chemical behavior in aqueous systems are strongly correlated. Because of the ubiquitous presence of iron in the Central Area aquifer, iron plays a dominant role in the behavior of arsenic at the Wells G and H site and is likely to be the dominant solubility control. In oxidized domains, ferric iron (Fe[III]) predominates and tends to precipitate from solution as iron oxyhydroxides (FeOOH). There are various physical forms of FeOOH, including

crystalline. However, the amorphous FeOOH solids tend to predominate when initially formed (precipitated). Amorphous FeOOH solids have a high specific surface area, and therefore can serve as sink of arsenic in oxidized aqueous systems, such as the Aberjona River. If redox conditions were present that allowed the presence of significant quantities of FeOOH solids throughout the aquifer at the pH ranges typically observed in groundwater in the Central Area aquifer, migration of arsenic to municipal wells G and H, or other high-volume potable water supply well would be significantly attenuated. As previously noted, available soil data show that iron is plentiful in Central Area soil. In addition, the presence of iron hydroxide solids is supported by the previously discussed Eh-pH regime for the Central Area aquifer and Eh-pH predominance graphics for iron (see Figure 3). Note, too, that arsenic also sorbs to aluminum oxides ($Al[OH]_3$) and significant concentrations of aluminum (as high as 6,740 ug/L at MW-002S) were detected in Central Area and nearby groundwater. Manganese oxides also oxidize As(III) to As(V), which is more likely to sorb (Oscarson et al, 1981).

The kinetics of arsenic transformation are important, as are the ORP conditions necessary to drive the transformation. The kinetics of arsenic transformation are slow. For example, the transformation of As(III) to As(V) under the imposition of oxidizing conditions can take days (Cherry et al, 1979). Therefore, species of arsenic in dis-equilibrium with their surroundings can exist for some time. The transformation is not instantaneous. Also, the speed with which As(V) is released from FeOOH, under the imposition of reducing conditions varies with the form of FeOOH solid. As(V) adsorbs readily to FeOOH. Amorphous FeOOH solids dissolve more quickly, and thus release their load of arsenic more rapidly than crystallized FeOOH due to thermodynamic differences. In addition, the reduction of FeOOH is facilitated by microbial mediation, which in turn is driven by the metabolism of organic matter. If conditions are not conducive to microbial mediation, then FeOOH transformation, and thus arsenic release, will be attenuated. As later discussed, conditions in medium and deep overburden in the Central Area aquifer are not conducive to biological activity, due in part to low concentrations of natural organic matter.

In oxidizing environments, arsenic exists primarily as oxyanions (hard anions that contain oxygen) and are immobile. In addition to iron oxyhydroxides, arsenic will also adsorb to clays, aluminum hydroxides, manganese compounds, and organic material at acidic and neutral pHs. Arsenic can be reduced from higher to lower valence states by organic matter, divalent metals, and dissolved sulfide. In sulfidic conditions, arsenic can form aqueous arsenic-sulfide complexes called thioarsenates (Wilkin et al, 2003, Rader et al, 2004)

5.1.2.2 Arsenic in the Central Area Aquifer

Within the glacial aquifer in the Central Area, measured ORP readings were never more than mildly reducing (-94 mV to -168 mV), with areas of groundwater exhibiting oxidizing conditions ranging as high as 270 to 556 mV. Under these conditions the less mobile and more highly sorbed As(V) can be formed. Figure 5 presents the site Eh-pH regime for arsenic superimposed on an Eh-pH diagram for aqueous arsenic species. Under the Eh-pH conditions generally present in the Central Area Aquifer, the majority of arsenic is expected to be present as the less mobile, more readily adsorbed As(V) species.

EPRI (2003) conducted soil leaching experiments with arsenic and determined that K_d values ranged from 0 to 664.2 milliliters per gram (mL/g). The best predictor of arsenic content in the leachate was soil grain size, and iron and aluminum content. K_d increased with decreasing grain size (about 3 orders of magnitude or more from sand to clay).

Hemond (1995) indicated that in the Aberjona watershed the pattern of arsenic contamination in the sediment does not seem to correlate to any probable distribution of sources; however, a strong relationship with the fineness of sediment texture and with organic matter content (which themselves are correlated) was observed. Naturally occurring levels of arsenic in soils are in the range of 0.4 to 40 mg/kg for soils with no geologic or anthropogenic arsenic inputs.

Studies by Keon et al (2000) of the arsenic rich and sulfidic peat sediments from the Aberjona watershed indicated that most of the arsenic is strongly adsorbed, with the remainder likely coprecipitated with iron oxyhydroxides. In contrast, riverbed sediments five feet from the wetland samples show a different arsenic distribution: arsenic is distributed among pools including crystalline sulfides, recalcitrant arsenic oxides, strongly adsorbed As, and ionically adsorbed arsenic (Keon et al 2000).

Keon et al (2001) employed a sequential extraction technique to differentiate various "pools" of solid phase arsenic in sulfidic Aberjona River sediments and iron-reducing wetland peat, and sought to quantify mobile arsenic and discern arsenic associations with redox-dependent solid phases (e.g., iron oxyhydroxides and sulfides). Keon et al (2001) suggests a variety of geochemical processes that might liberate the variously bound fractions of arsenic in sediments. These potential arsenic mobilizing processes include, for example increases in the ionic strength of the aqueous medium, which could occur from road salting, or removal of arsenic complexed with iron oxides or humic acids due to phosphate-laden runoff (e.g., agricultural drainage or lawn fertilization). Keon et al (2001) also attempted to discern the mobility of arsenic coprecipitated with manganese oxides and amorphous iron oxyhydroxides, and arsenic adsorbed to particle sulfates using a hydrochloric acid extraction (Keon et al, 2001). However, the Keon et al (2001) extraction procedure did not account for re-precipitation of arsenic as an arsenic sulfide (Ford, 2004c). Wilkin and Ford (2002) note that with sediments containing sulfide minerals, the hydrochloric acid extraction introduces artifacts to the partitioning scheme. Consequently, Keon et al (2001) may not accurately describe the potential mobility of arsenic associated with coprecipitated manganese oxides and amorphous iron oxyhydroxides, as well as arsenic sorbed to particulate sulfate.

Arsenic in the Aberjona River and downstream lakes was analyzed by Hemond (1995). Hemond found that in the Mystic Lakes downstream of Wells G and H, the arsenic concentrations varied seasonally from 1 ug/L during the winter months to 1.3 ug/L during the summer months. Hemond (1995) found that arsenate concentrations in the anoxic hypolimnion actually increased during the summer months and the arsenite increased from one-quarter of the total arsenic in the water during the colder months to one-half of the total between May and October in the mixed layer (epilimnion) despite that fact that the upper water remained well oxygenated. He indicated that these counter intuitive results are evidence that kinetic considerations govern arsenic speciation during much of the year. He also found that organic arsenic monomethylarsonic acid (MMAA) and dimethylarsinic acid (DMAA) followed a temporal and spatial pattern similar to

that of arsenate, reaching maximum concentrations respectively in the excess of 0.3 ug/L and about 0.1 ug/L. Hemond (2001) also indicates that the Nitrogen Cycle impacts arsenic speciation. Nitrification represents a major hypolimnetic oxygen demand following the onset of seasonal stratification. Nitrate controls the redox potential of the bottom waters and is responsible for reoxidizing ferrous iron (Fe[II]) as it diffuses from the lake sediments into the hypolimnion. Studies by Ahmann et al (1994) have isolated two new species of bacteria from Wells G and H wetland that reduce arsenic, one can also precipitate orpiment (As_2S_3) and have a key role in controlling the mobility of arsenic in the watershed.

Other site-specific factors will influence the geochemical behavior and mobility of arsenic. As discussed herein, the mildly reducing to oxidizing aqueous environment of the Central Area aquifer should favor the formation of less mobile As(V), and Fe(III), which provides an adsorption surface for arsenic. The formation of As(V) should further be favored along flow paths by the induced infiltration of demonstrably well oxygenated river water under pumping conditions. Induced infiltration of relatively low concentration river water should also act to dilute contaminant concentrations in the aquifer.

Note that Ford (2004b) conducted oxic leaching tests to assess arsenic release under oxic conditions using a leaching solution with a chemical composition consistent with surface water. Sediment pore water measurements and oxic leach test results indicate that sediments may be a long-term source of arsenic to groundwater at Wells G and H. Arsenic concentrations released from sediments collected from the Wells G and H wetland ranged from 23 $\mu\text{g/L}$ to 341 $\mu\text{g/L}$. Ford (2004b) notes, however, that the greatest risk of arsenic release from sediments occurs under reducing conditions. Under a pumping scenario where aqueous geochemical conditions become increasingly dependent on the influence of induced surface water infiltration, the aquifer becomes more oxidized; as a result, steady concentrations like that found in the batch oxic leach tests are unlikely to be sustainable. This outcome is dependent, however, on the degree of mixing of various water sources within the aquifer and would be influenced by microbial activity (Ford 2005a).

Overall, as discussed later in this document, water quality data collected during operation of Wells G and H, and later data collected during remedial investigations and the USGS pump test, do not trigger criteria noted as indicative of high arsenic groundwater by Smedly and Kinniburgh (2004).

Groundwater data collected from the Central Area aquifer and nearby areas on behalf of EPA in 2002 using low stress (low-flow) purging protocols set forth by EPA Region I detected arsenic in 76 out of 77 groundwater samples. The highest concentrations tended to be located in the Central River Valley. Fourteen out of the 77 groundwater samples exceeded the current 10 $\mu\text{g/L}$ arsenic MCL. In addition, the highest concentrations were also found in the shallow overburden wells. Comparisons between data from historical non-low-flow and the aforementioned 2002 low-flow sampling suggest that non-low-flow samples techniques may have contributed to elevated arsenic concentrations due to aquifer solids entrainment in the samples. However, since arsenic does not partition as strongly to aquifer solids as chromium and lead, which partition very strongly to aquifer solids (see Sections 5.1.3 and 5.1.4), a portion of the reductions in arsenic concentrations could represent natural attenuation.

5.1.3 Chromium

Chromium has been identified as a sediment contaminant in the Aberjona River. EPA determined that a significant portion of the chromium found in sediments was chromium III (greater than 99-percent) (M&E, 2004). The Aberjona River Study did not identify chromium in sediment as a risk driver for human health; however, chromium in sediment of the river and the 38-acre wetland does pose a risk to ecological receptors such as the mallard duck and muskrat. The following summary of chromium fate and transport properties will aid in the interpretation of available groundwater chromium data and related geochemical/water quality data.

Chromium is an inorganic chemical that occurs in the earth's crust and is released to soil and groundwater from natural and anthropogenic sources. Chromium can be transported from soil by wind erosion or runoff, or can leach into the subsurface.

In oxidizing environments, these chromium compounds primarily exist as oxyanions (hard anions that contain oxygen) and are relatively mobile. However, they can be absorbed by clays, iron, oxyhydroxides, aluminum hydroxides, manganese compounds, and organic material at acidic and neutral pHs. Chromium can be reduced from higher to lower valence states by organic matter, divalent metals, and dissolved sulfide.

Chromate (Cr[VI]) species predominate and are highly mobile in oxidized systems, except for their tendency to be adsorbed, particularly by Fe(III) and Mn-oxides below pH 8. Chromate is in general weakly complexed. In contrast, Cr(III) dominates in reducing environments and forms strong, hard-acid complexes. Cr(III) complexes with hydroxyl, sulfate, organic ligands, and other species, which increases its stability and increases the solubility of Cr₂O₃ (chromium oxide – the solid form). Organic matter, Fe(II), and hydrogen sulfide (H₂S) can reduce Cr(VI) to Cr(III). Manganese oxides in soil can adsorb Cr(III) and oxidize it to Cr(VI). Adsorption of chromate (C₂O₄) by hydrous ferric oxide decreases with increasing pH, whereas Cr(III) adsorption increases as the pH rises (ERG, 2003).

In the general Eh-pH regime of the Central Area aquifer (see Figure 6), chromium is expected to be in the Cr(III) state and will likely exist as Cr₂O₃ solid. In addition, K_d values for Cr (III) suggest that chromium will be relatively immobile in Central Area groundwater. The following table compares the K_d values derived from ERG (2003) for arsenic and chromium at various pHs. The K_d values for Cr(III), the chromium species expected to predominate in the Central Area aquifer, are many orders of magnitude greater than those of As(V), the least mobile of the two primary arsenic species expected in Central Area groundwater in the Central Area aquifer. Thus, chromium is expected to be relatively immobile.

Soil Water Adsorption Partitioning Coefficient (Kd)			
Element	pH 4.9	pH 6.8	pH 8.0
As(V)	25	29	31
Cr (III)	1,200	1,800,000	4,300,000
Cr (VI)	31	19	14

Notes:

*Kd = Cs/Cw where Cs = concentration adsorbed into soil; Cw = concentration in water units = mL/g
Kd units = mL/g

5.1.4 Lead

Lead has been identified as a sediment contaminant in the Aberjona River. The Aberjona River Study did not identify lead in sediment as a risk driver for human health; however, lead in sediment does pose a risk to ecological receptors such as mallard ducks and muskrats. The following summary of lead fate and transport properties will aid in the interpretation of available groundwater lead data and related geochemical/water quality data.

Lead is present in the earth's crust and may be released from weathering processes and from anthropogenic sources. This element is a borderline hard/soft cation that forms insoluble metal sulfides in anaerobic environments. Lead generally exhibits little mobility in groundwater systems and tends to sorb and be transported in water primarily with suspended colloidal particles (ERG, 2003). The concentration of lead in groundwater is usually controlled by adsorption or coprecipitation with iron, manganese, and aluminum. Lead is relatively immobile in all matrices due to its strong tendency to be sorbed by iron and manganese oxides and the insolubility of many lead minerals. Lead is effectively removed from water by adsorption to organic matter and clay minerals, precipitation as insoluble salt, and the reaction with hydrous iron and manganese oxide.

Lead is relatively immobile in soils, sediments, and groundwaters. This is reflected by its strong tendency to be adsorbed by Fe and Mn oxides under oxidizing conditions, but also the insolubility of a number of lead minerals including lead hydroxycarbonate, which limits lead concentrations in some public water systems, and pyromorphite ($Pb_5(PO_4)_3Cl$), which controls lead concentrations in some soils adjacent to highways affected by road salt and leaded gasoline exhaust. Lead sulfides are also very insoluble at low Eh, even at low metal and sulfide concentrations (ERG, 2003). Mechanisms for sorption of lead by soils and sediments are ion exchange, specific adsorption, co-precipitation with hydrous oxides, and incorporation into cationic lattice sites in crystalline sediments (Bodek et al, 1988). The following table derived from Bodek et al (1988) summarizes typical values of sorption constants (Langmuir constants) for lead on materials such as clays, Fe-oxides, Mn-oxides, organic matter, soils and sediments. The A_m and $\log K_L$, which represent the maximum adsorption capacity of the solid and the sorption constant related to the binding energy of sorbate, respectively, demonstrate the predominance of Fe and Mn oxide as a sink or solubility control for lead under oxidizing conditions.

Langmuir Constants	Clays	Fe Oxides	Mn Oxides	Organic Matter	Soils	Sediments
A_m	~40	2400	2700	300-500	31	20
Log K_L (log M^1)	3.5	~4	3.8	~4	4.5	4.9

Notes:

A_m = maximum adsorption capacity of solid (ugmol/g) – a Langmuir constant

Log K_L = Sorption constant related to binding energy of sorbate – a Langmuir constant

In the general Eh-pH regime of the Central Area aquifer (see Figure 7), lead is expected to be in the soluble Pb(II) state, but should be readily adsorbed given the extent of Fe and Mn oxides in the Central Area aquifer and the adsorption tendencies expressed by the Langmuir constants.

Groundwater data collected from the Central Area aquifer and nearby areas on behalf of EPA in 2002 using low stress (low-flow) purging protocols set forth by EPA Region I showed that lead was detected in Central Area groundwater at concentrations generally an order of magnitude lower than the EPA Safe Drinking Water Act (SDWA) Action Level of 15 ug/L. The highest concentration detected during the 2002 low stress (low flow) sampling (5.2 ug/L at MW-002M) was lower than the action level by a factor of 3. Historical analytical data between 1979 and 1991 detected lead on a limited number of events. One sample collected on 1/25/81 from Well G anomalously detected lead at a concentration of 270 ug/L. Historical groundwater data from samples collected with the non low-flow sampling techniques may have artificially elevated metals concentrations due to aquifer solids entrainment (turbidity). Comparisons between data from historical non-low-flow and more recent low-flow sampling discussed later in this document support this hypothesis. Low-flow data are considered more representative of actual conditions. No prior or subsequent monitoring detected lead in Wells G or H except at the end of the USGS aquifer pump test on 1/3/1986. Lead was detected in aquifer pump test effluent from Wells G and H at concentrations of 90 ug/L and 80 ug/L, respectively. However, samples collected 10 days prior during the aquifer pump test did not detect lead at a laboratory reporting limit of 50 ug/L. Later sampling unrelated to the pump test conducted on 8/21/91 did not detect lead at a laboratory reporting limit of 5 ug/L. The erratic detections are contrary to the limited transport behavior of lead and could be artifacts of sampling technique (none of the historical data were based on low-flow sample collection) or could reflect another source of contamination. For example, brass or bronze parts on the pump could serve as a source of detectable lead (WWJ, 1994).

5.1.5 Copper

Copper was also identified as a sediment contaminant in the Aberjona River. Although the Aberjona River Study did not identify copper in sediment as a risk driver for human health, copper was found to pose a risk to ecological receptors, specifically, mallard ducks. The following summarizes salient properties, and fate and transport properties of copper that are important in evaluating its potential for contamination of Wells G and H.

Copper is strongly adsorbed by organic matter, Fe(III) and Mn oxides, and secondarily by clays (ERG, 2003). Bodek et al (1988) state that copper exceeds lead in its tendency to be sorbed to solids. Sorption is probably the most important controlling mechanism in determining copper

mobility in the environment. Organic complexation, specific adsorption, precipitation, and ion exchange are important mechanisms for sorption of copper onto soils and sediments. Studies have shown that sorption of copper on hydrous iron and manganese oxides is due to co-precipitation of copper in the iron and manganese oxide lattice (Bodek et al, 1988). The following table derived from Bodek et al (1988) summarizes typical values of sorption constants (Langmuir constants) for copper on materials such as clays, Fe-oxides, Mn-oxides, organic matter, soils and sediments. The A_m and $\text{Log } K_L$, which represent the maximum adsorption capacity of the solid and the sorption constant related to the binding energy of sorbate, respectively, demonstrate the predominance of Fe and Mn oxide as a sink or solubility control for copper. Also tabulated is the K_d for copper on clay, Fe oxides and Mn oxides. The K_d values for Fe and Mn oxides are consistent with those of Cr (III), which is not very mobile.

Adsorption Properties of Copper						
Sorption Constants	Clays	Fe Oxides	Mn Oxides	Organic Matter	Soils	Sediments
A_m	6	130	1670	200	28	34
$\text{Log } K_L$	5	5	4	5	5	5
K_d	3	3600	8000	--	--	--

Notes:

- A_m – maximum adsorption capacity of solid (ugmol/g) – a Langmuir constant
- $\text{Log } K_L$ – Sorption constant related to binding energy of sorbate – a Langmuir constant
- K_d – Distribution coefficient (mL/g)
- No value provided in Bodek et al (1998)

Copper is insoluble in reduced environments, where it will precipitate as a sulfide. Copper is insoluble above pH 7–8 (because of the precipitation of tenorite, CuO) and in the presence of abundant carbonate of Cu(II) carbonate minerals. Cu(II) forms strong humate complexes, so that in soils >99.9-percent may be complexed (ERG, 2003).

In the general Eh-pH regime of the Central Area aquifer (see Figure 8), copper will largely exist in a precipitated phase, and available soluble Cu(II) species should be readily adsorbed to Fe and Mn oxides given the extent of Fe and Mn oxides in the Central Area aquifer and the adsorption tendencies expressed by the Langmuir constants and distribution coefficient. Copper groundwater data collected on behalf of EPA in 2002 using low stress (low flow) purging protocols set forth by EPA Region 1 showed that copper was detected in Central Area groundwater at concentrations approximately 3 orders of magnitude lower than the EPA drinking water action level of 1,300 ug/L.

5.1.6 Mercury

Mercury (Hg) has been identified as a sediment contaminant in the Aberjona River. The Aberjona River Study did not identify mercury in sediment as a risk driver for human health; however, mercury in sediment does pose a risk to ecological receptors such as mallard ducks and muskrats. The following summarizes salient fate and transport properties of mercury that are important in evaluating its potential for contamination of Wells G and H.

Mercury can exist in three oxidation states (elemental mercury Hg(0), mercurous mercury Hg(I), or mercuric mercury Hg(II)). Mercury forms strong complexes with chloride ion (Cl⁻), which may dominate the solution chemistry of this metal. Mercury forms even stronger complexes with other halogens such as bromide and iodide. Mercury is highly insoluble in reduced environments, where it can precipitate as a metal or as a sulfide. Hg(II) is usually complexed—in pure water as Hg(OH)₂⁰, and at chloride concentrations typical of fresh waters (<10E-2 mol/kg) as HgCl₂⁰. Hg(II) forms strong complexes with humic materials, so that in soils >99.9-percent of mercury may be complexed (ERG, 2003).

Mercury is biologically methylated only in environments low in sulfide. The stable methyl mercury species in fresh waters is CH₃HgOH. The methyl mercury cation, CH₃Hg⁺, complexes with ligands containing carbon (C), phosphorous (P), oxygen (O), nitrogen (N), and the halogens, and forms very stable complexes with sulfur-containing ligands. In oxidized, fresh waters the Hg(II) methyl hydroxo and methyl chloro complexes dominate (ERG, 2003).

Mercury is strongly adsorbed by organic matter, Fe(III) and Mn oxides, and secondarily by clays (ERG, 2003). Sorption of mercury is very fast and practically irreversible (Bodek et al., 1988). By inhibiting mercury sorption, Hg-Cl complexing helps to mobilize the metal. Dimethyl mercury is very insoluble in water and tends to be volatilized from soils (ERG, 2003)

In the general Eh-pH regime of the Central Area aquifer (see Figure 9), mercury will largely exist in low solubility phases, and available soluble mercury species should be readily adsorbed to Fe and Mn oxides given the extent of Fe and Mn oxides in the Central Area aquifer. In addition, mercury groundwater data collected on behalf of EPA in 2002 using low stress (low flow) purging protocols outlined by EPA Region 1 showed that mercury was rarely detected in Central Area groundwater. Detected concentrations, and laboratory reporting limits for non-detect results, were typically more than an order of magnitude lower than the EPA MCL of 2 ug/L (generally 0.1 ug/L).

Mercury was anomalously detected at the conclusion of the USGS 30-day pump test in Wells G and H at concentrations of 3.8 ug/L and 4.4 ug/L, respectively, which exceeds the mercury MCL of 2 ug/L. Samples collected approximately 10-days prior to the conclusion of the pump test did not detect mercury at a laboratory reporting limit of 0.2 ug/L. The mercury detections at the end of the pump test are anomalous given that groundwater in the surrounding aquifer does not contain mercury at significant concentrations.

6.0 ANALYSIS

Arsenic is the focus of this analysis, although the potential impacts of other metals, including chromium, lead, iron and manganese are also considered. The probable fate of copper and mercury are also discussed. The influence of geochemical parameters such as ORP, DO, and sulfate are included.

6.1 Preface to Analysis

The most straightforward method for evaluating Aberjona River surface water and sediment impacts on potable water development in the Central Area (OU-2) aquifer would be to assess water quality of the former municipal wells G & H during operation. Unfortunately, arsenic data collected during the period of the use of the wells as a potable water supply are not available. However, other secondary data are available to allow a supposition regarding whether Aberjona River water and sediment arsenic concentrations could have an impact on potable water development. The assessment criteria used to evaluate Aberjona River impacts on potable water development include the following:

- Water quality data from Wells G and H from a period shortly after pumping ceased and proxy data from the period of Wells G and H operation (pre-1979) that indicate whether Wells G&H were likely impacted by arsenic; Water quality data for other pumped wells in the proximity of Wells G and H;
- Wells G and H water quality data (non-arsenic parameters) compared to other aquifers where elevated arsenic is present;
- Mobility of arsenic;
- Arsenic concentrations in the Wells G and H area;
- Hydrologic factors controlling the migration of arsenic;
- Influence of peat deposits;
- Measurement of human hair samples; and
- Other issues.

6.2 Water Quality Data for Wells G and H

Wells G and H were used intermittently as a potable water supply, primarily for emergency water demand, by the City of Woburn. The wells were pumped continuously in the summer months at rates of 700-800 gpm and 400 gpm, respectively (USGS, 1987).

Well G was constructed in late 1964. It is screened from a depth of approximately 75 to 85 feet below ground surface. Well H was constructed in 1967. It is screened from a depth of approximately 78 to 88 feet below ground surface. In both wells, the gravel pack extends

upward to within 10 feet of the surface, above which the casing is surrounded with grout or concrete.

Wells G and H were constructed in soils consisting of an upper layer of peat underlain by sand and gravel to a depth of about 80 to 90 feet. When the wells were first pumped, coliform contamination was identified in the wells. The coliform problem was reportedly "corrected by excavating all of the organic soil or peat for 30 feet around the wells and replacing it with clean bank-run sand and gravel" (Dufresne-Henry, 1978). The well logs indicate that the peat was 5 feet thick at Well G and 17 feet thick at Well H.

Water quality data collected by the City of Woburn during the operation of municipal Wells G and H are available from 1967, 1968, 1973, and 1974. Unfortunately, this data set does not include arsenic and anthropogenic metal contaminant data of interest to this analysis. However, this water quality data is useful for evaluating the redox conditions of the aquifer during pumping as well as to provide comparison to indicator parameters for high arsenic groundwater discussed below.

Arsenic and metals data are available for Wells G and H from sampling conducted on September 24, 1979 (approximately 4 months after shutdown) contained 2.0 and 1.5 ug/L of arsenic, respectively. In six subsequent sampling rounds since 1981, arsenic was nondetect at laboratory reporting limits of 5 to 10 ug/L. In 1986, a 30-day combined pumping test of Wells G and H was performed by the USGS to determine the area of influence and the zone of contribution to the two wells. The wells were pumped at rates of 700 and 400 gpm, respectively. During the pump test, groundwater samples were collected for chemical analysis by other parties at approximately 20 days into the test and at the end of the test. Groundwater samples collected at the 20 day point by GeoEnvironmental Consultants and at the end of the pump test by Weston Geophysical Corporation did not detect arsenic at laboratory reporting limits of 10 ug/L and 5 ug/L, respectively (ETC, 1986; Tighe & Bond, 1986). It is possible that the 30-day USGS pumping test may not have been representative of long-term conditions. However, the other constituents present in the groundwater such as manganese, pH and hardness (and others) were at levels similar to those detected at Wells G and H during operation (Table 1), which suggests that the arsenic results are representative. In any case, the results of all these post-shutdown chemical analyses of Wells G and H suggest that arsenic contamination levels would have been relatively low during the long-term operation of the wells and would probably have met current drinking water standards for arsenic (2006 MCL of 10 ug/L).

Table 1. Comparison Between Historical Wells G and H Water Quality During Pumping to USGS Pump Test Groundwater Analyses Data

Constituent	Well G During Municipal Pumping	Well G During USGS Pump Test	Well H During Municipal Pumping	Well H During USGS Pump Test
Iron	0.01-0.12	<0.03	0.01-2.4	<0.03
Manganese	0.02-0.55	0.36	0.59-2.16	0.85
pH	6.2-6.9	6.5	6.3-6.6	6.5
Hardness	79-250	122	188	138
Nitrate as N	0.4-4.4	1.67	0.01-4.9	1.57
Chloride	37-120	72.5	107	86
Conductivity	500-700	446	-	487
Copper	<0.01-0.05	<0.02	-	-

Notes

Concentration in mg/L except conductivity in umhos/cm and pH in pH units.

- Data not available/reported

Sources: Dufresne-Henry (1978), Tighe and Bond (1986) and historical Wells G and H operational chemical analysis data received from Anna Mayor on City of Woburn Department of Public Works well testing (Mayor, 2003).

Just before the start of the USGS aquifer pumping test, measurements of streamflow indicated that the Aberjona River was gaining water at a rate of about 770 gpm between Olympia Avenue and Salem Street. Within one day after starting the test, that same part of the river was losing water at a rate of about 140 gpm. Gradients had been reversed at that time, causing groundwater that would have discharged to the river and seepage from the river and wetlands to move toward the wells. Near the end of the USGS pumping test, when Wells G and H had been pumped at 1,100 gpm combined for 25 to 30 days, the length of river near the wells was losing water at a rate of 550 to 650 gpm. The cone of depression extended beneath this entire length of the river, so the potential existed throughout this area for water to move downward into the aquifer from the river and the wetlands. See Figure 10 for an illustration of the Wells G and H drawdown (zone of influence) after 30 days of pumping (January 3, 1986).

If large-scale groundwater withdrawals were again made from the aquifer at or near Wells G and H, the interactions between the river and the aquifer would probably be similar to those recorded by the USGS in 1987, and seepage from the river into the aquifer would be significant. The Aberjona River currently has a dissolved arsenic level of about 6 ug/L during baseflow at sampling station TT#5, located downstream of the Wells G and H at the Salem Street bridge (TTNUS, 2002), which would be the dissolved arsenic concentration of the water as it entered the underlying sediment. Particulate arsenic in the river water would most likely be filtered out as the water passed through the sediments and the aquifer.

The dissolved arsenic concentration of the river water would most likely be modified to varying degrees, depending on the subsurface conditions along its flow path down into the aquifer.

In addition, the induced infiltration of oxygenated (DO ranging seasonally from 4.2 to 11.8 mg/L) and strongly oxidizing (ORP ranging from +170 mV to +485 mV) surface water from the

stretch of the Aberjona River near Wells G and H during pumping should shift the aqueous chemical equilibrium of the aquifer to more oxidizing conditions along flow paths. The equilibrium of aqueous arsenic would be expected to increase the predominance of less mobile As(V). In addition, increased formation of iron oxyhydroxide solids would also be expected, to which As(V) very strongly sorbs. Note, however, the arsenic redox equilibrium can be slow to achieve, as shown by Cherry et al (1979) and others. In addition, conditions are expected to be more reducing at the sediment/groundwater interface.

The upland groundwater component of the flow toward the wells could also be modified as it moves through the aquifer, although it would not migrate through potentially arsenic-rich riverbed sediments. Unless much of the infiltrating river water developed much higher dissolved arsenic concentrations as it passed through the sediment and peat, the average of the surface water and groundwater concentrations would yield arsenic concentrations less than 10 ug/L, but a detailed groundwater contaminant transport model would be required to confirm this hypothesis. Based on the low hydraulic conductivity of the peat and the high hydraulic conductivity of the glacial stratified drift sediments, the water originating in the peat would be diluted more than 1,000 fold. Although the above-referenced arsenic-bearing water sources may be characterized by low arsenic concentrations on average, due to geochemical and transport dynamics, the source water may become either enriched or lose arsenic due to geochemical reactions in the subsurface due to the partitioning behavior of arsenic and the dissolution/precipitation behavior of iron oxyhydroxides. As later discussed, the dynamics of the well operation could change these equilibrium values; however, sufficient data are not now available to show that high arsenic concentrations would result in the aquifer under the high-capacity groundwater withdrawal scenario.

6.3 Water Quality Data for Other Pumped Wells in the Proximity

Several wells in the proximity of Wells G and H were pumped for many years and in some cases are still being pumped. If the water quality in these wells is low in arsenic, then other wells placed nearby for potable water use would also be low in arsenic, provided the hydraulics are relatively consistent and the screen depths are similar. Nearby pumped wells include the two J.J. Riley Tannery wells located along the southwest boundary of the Wells G and H Site (Riley Wells), and the Atlantic Gelatin well field located southeast of the Wells G and H site on the Woburn/Winchester line.

6.3.1 J.J. Riley Tannery Water Supply Wells

The two Riley Wells (S47 and S46) were installed in 1945 and 1958, respectively (Delaney and Gay, 1980) and the J.J. Riley Tannery operated until 1988 (TTNUS, 2003). The wells each had rated capacities between 500 to 750 gpm, which is comparable to the capacities of Wells G and H. The well depths were between 35 and 51 feet, which is about half as deep as Wells G and H (Delaney and Gay, 1980). The results of arsenic analysis in both Riley wells from the only sampling event conducted during their period of operation (1980) were non-detect at a laboratory reporting limit of 10 ug/L (GeoTrans, 1994).

6.3.2 Atlantic Gelatin Well Field

Atlantic Gelatin is permitted to withdraw up to 1 million gallons per day (MGD) from 4 operational wells. In the 1950s these wells were pumped at a rate of about 1.5 MGD (Lubker and Halpin, 1951). The groundwater is used for process wash water. Wastewater is discharge to the MWRA sanitary sewer system (TTNUS, 2003). The Atlantic Gelatin wells were installed between about 1935 and 1958, and were between 62 to 156 feet deep. The rated capacities of the wells ranged from 372 to 1,000 gpm (Delaney and Gay, 1980). No testing of water quality for arsenic has been conducted.

6.4 Comparison to Other Aquifers

A comprehensive review of arsenic contamination in groundwater was compiled by Smedley and Kinniburgh (2004). They determined concentration ranges for 7 water quality parameters that are indicative of high arsenic concentrations. Evaluating the Wells G and H data in comparison to these parameters showed that unfavorable conditions were present in Well G (all 7 parameters unfavorable) and Well H (5 of 7 parameters unfavorable) for the presence of high arsenic levels. The parameters evaluated included iron, manganese, alkalinity, chloride, sulfate, nitrate, and pH. Table 2 presents the results of the indicator parameters.

Analyte	Favored Range for High Arsenic (mg/L) ¹	Well G			Well H			Finding	
		Avg	Low	High	Avg	Low	High	Well G	Well H
Iron	>0.2	0.045	0.000 ²	0.210	0.631	0.010	2.400	Unfavorable	Favorable
Manganese	>0.5	0.410	0.020	1.170	1.123	0.200	2.160	Unfavorable	Favorable
Alkalinity	>500	50	32	60	50	47	56	Unfavorable	Unfavorable
Chloride	<60	69	19	185	90	25	116	Unfavorable	Unfavorable
Sulfate	<1	89	22	143	114	80	150	Unfavorable	Unfavorable
Nitrate	<1	3	0.4	21	3.8	0.03	32	Unfavorable	Unfavorable
pH	>7 SU	6.6	6.2	7.3	6.5	6.3	6.7	Unfavorable	Unfavorable

Notes:

¹ From Smedley and Kinniburgh, 2004.

² As posted in Mayor 2003.

SU – Standard units for potential of hydrogen (pH)

Wells G and H data sources: Mayor 2003; GeoTrans, 1994; and Dufresne-Henry, 1978

Table 2 includes some of the characteristic chemical features of high arsenic groundwaters noted by Smedley and Kinniburgh (2004), such as high iron concentrations (>0.2 mg/L), high manganese concentrations (>0.5 mg/L), and low concentrations of chloride (<60 mg/L), sulfate (<1 mg/L), and nitrate (<1 mg/L), as well as pH values near or greater than 7. Other chemical characteristics that are indicative of the relative presence of arsenic include bicarbonate and fluoride; however, comparative data from Wells G and H are not available for these parameters and therefore were not included in Table 2. Strong direct correlations between arsenic and iron concentrations in aqueous systems have been noted in the scientific literature and in work

conducted by EPA at the Industri-Plex Superfund Site. Consequently, high aqueous iron concentrations are expected to be indicative of the potential for high concentrations of arsenic. Manganese oxides also undergo reductive desorption and dissolution, and so could contribute to the arsenic load of groundwaters in the same way as iron. As shown in Table 2 iron and manganese levels at Wells G and H exhibit both favorable and unfavorable indications for a presence of arsenic; however, the possibility for arsenic presence needs to be assessed in view of all the parameters. Total arsenic has also been found to be positively correlated with alkalinity and pH. High salinity (monitored by chloride measurements) in fresh water aqueous systems associated with high rates of evaporation in some regions can be associated with high arsenic concentrations in groundwater. However, chloride is not expected to be a significant indicator in the non-arid conditions characteristic of the Woburn-area. Low sulfate concentrations can be indicative of strongly reducing conditions, and high arsenic concentrations are often associated with strongly reducing conditions. Sulfide mobilizes arsenic due to the formation of thioarsenite complexes (Wilken et al, 2003; Rader et al, 2004). Nitrate levels are also indicative of redox conditions, with low nitrate levels indicative of highly reducing conditions favorable to the presence of high arsenic concentrations. Overall, as shown in Table 2, the average geochemical conditions recorded in Wells G and H exhibit conditions unfavorable to the presence of high arsenic concentrations.

6.5 Arsenic Mobility

Hinkle and Polette (1999) reported that two categories of processes largely control arsenic mobility in aquifers:

- Adsorption and desorption reactions; and
- Solid-phase precipitation and dissolution reactions.

Arsenic adsorption and desorption reactions are influenced by changes in pH, redox (reduction/oxidation) reactions, presence of competing anions, and solid-phase structural changes at the atomic level. Solid-phase precipitation and dissolution reactions are controlled by solution chemistry, including pH, redox state, and chemical composition.

6.5.1 Adsorption and Desorption Processes

Arsenic is a redox-sensitive element that gains or loses electrons in redox reactions, and may be present in a variety of redox states. Arsenate and arsenite are the stable forms of arsenic solution in water (Hem, 1985). As(V) generally predominates under oxidizing conditions. As(III) predominates when conditions become sufficiently reducing. Under the pH conditions of most groundwater, As(V) is present as the negatively charged oxyanions $\text{H}_2\text{AsO}_4^{-4}$ or HAsO_4^{-2} , whereas As(III) is present as the uncharged aqueous species H_3AsO_3 (Hem, 1985). The strength of adsorption and desorption reactions between these different arsenic species and solid-phase surfaces in aquifers varies, in part, because of these differences in charge. Differences in species charge affect the character of electrostatic interactions between species and surfaces.

As(V) and As(III) adsorb to surfaces of a variety of aquifer materials, including iron oxides, aluminum oxides, and clay minerals. Adsorption and desorption reactions between As(V) and

iron oxyhydroxide solid surfaces are particularly important controlling reactions because iron oxides are widespread in the hydrogeologic environment as coatings on other solids, and because As(V) adsorbs strongly to iron oxyhydroxide surfaces in acidic and near-neutral-pH water (Pierce and Moore, 1982). However, desorption of As(V) from iron-oxide surfaces becomes favored as pH values become alkaline (Pierce and Moore, 1982; Masscheleyn et al, 1991b). The pH-dependence of As(V) adsorption to iron-oxide surfaces appears to be related to the change in iron-oxide net surface charge from positive to negative as pH increases above the zero-point-of-charge (pH at which the net surface charge is equal to zero) of about 7.7 for goethite, a crystalline iron oxide (Stumm and Morgan, 1996), or 8.0 for ferrihydrite, an amorphous iron oxide (Dzombak and Morel, 1990).

Iron oxyhydroxide surfaces also adsorb As(III) but to a lesser degree than As(V). Both As(V) and As(III) adsorb to aluminum oxides and clay-mineral surfaces. However, these adsorption reactions appear generally to be weaker than is the case for As(V) adsorption to iron-oxide surfaces under typical environmental pH conditions (Manning and Goldberg, 1997). Nevertheless, pH-dependent adsorption and desorption reactions other than those between As(V) and iron oxyhydroxide surfaces may be important controls over arsenic mobility in some settings. As is the case for adsorption of As(V) to iron oxyhydroxide surfaces, adsorption of As(III) to iron oxyhydroxide surfaces tends to decrease as pH increases, experiencing a maximum at pH 7. Unfortunately, As(V) and As(III) adsorption and desorption reactions with other common surfaces are less well characterized, and apparently more complex than is the case for adsorption and desorption reactions with iron oxyhydroxide surfaces (Manning and Goldberg, 1997).

As a result of the pH dependence of arsenic adsorption, changes in groundwater pH can promote adsorption or desorption of arsenic. Because solid-phase diagenesis (water-rock interaction) typically consumes hydronium ion (H^+) (Stumm and Morgan, 1996), the pH of groundwater tends to increase with residence time, which, in turn, increases along groundwater flowpaths. Because iron oxyhydroxide surfaces can hold large amounts of adsorbed As(V), geochemical evolution of groundwater to high (alkaline) pH can induce desorption of arsenic sufficient to result in exceedances of the current EPA MCL in some environments (Robertson, 1989). The Central Area groundwater tends to be mildly acidic; therefore high alkalinity desorption is not expected to be a factor at Wells G and H.

Similarly, redox reactions can control aqueous arsenic solubility by their effects on arsenic speciation, and hence, arsenic adsorption and desorption. For example, reduction of As(V) to As(III) can promote arsenic mobility because As(III) is generally less strongly adsorbed than is As(V). Redox reactions involving either aqueous or adsorbed arsenic can affect arsenic mobility (Manning and Goldberg, 1997).

Arsenic adsorption also can be affected by the presence of competing ions. In particular, phosphate and arsenate have similar geochemical behavior, and as such, both compete for sorption sites (Hingston, et al, 1971; Livesey and Huang, 1981; Manning and Goldberg, 1997). Oxyanions in addition to phosphate also may compete for sorption sites. For example, Robertson (1989) suggested that correlation of arsenate with oxyanions of molybdenum, selenium, and

vanadium in groundwater of the Southwestern United States may be evidence for competitive adsorption among those oxyanions.

Finally, structural changes in solid phases at the atomic level also affect arsenic adsorption and desorption. For example, conversion of ferrihydrite to goethite or to other crystalline iron-oxide phases may occur gradually over time (Dzombak and Morel, 1990). Fuller et al (1993) demonstrated that as ferrihydrite crystallizes into goethite, the density of arsenic adsorption sites decreases. This decrease in density of adsorption sites can result in desorption of adsorbed arsenic. Structural changes in other solid phases may possibly affect arsenic mobility, but the role of such solid-phase structural changes on groundwater arsenic concentrations has received little attention to date (Hinkle and Polette, 1999).

6.5.2 Precipitation and Dissolution Processes

The various solid phases (e.g., minerals, amorphous oxides, organic carbon) of which aquifers are composed exist in a variety of thermodynamic states. At any given time, some aquifer solid phases will be undergoing dissolution, whereas others will be precipitating from solution. Arsenic contained within solid phases, either as a primary structural component of, or as an impurity in, is released to groundwater when those solid phases dissolve. Similarly, arsenic is removed from groundwater when solid phases containing arsenic precipitate from aqueous solution. As an example, because arsenic often coprecipitates with iron oxide (Moore, et al, 1988), iron oxide may act as an arsenic source (in the case of dissolution) or a sink (in the case of precipitation) for groundwater. Furthermore, solid-phase dissolution will contribute not only arsenic contained within that phase, but also any arsenic adsorbed to the solid-phase surface. The process of release of adsorbed arsenic as a result of solid-phase dissolution is distinct from the process of desorption from stable solid phases (Hinkle and Pollette, 1999).

The interplay of redox reactions and solid-phase precipitation and dissolution may be particularly important with regard to aqueous arsenic and solid-phase iron oxides and sulfide minerals. High concentrations of arsenic often are associated with iron oxides and sulfide minerals (Thornton, 1996). Iron oxides frequently dissolve under reducing conditions, but often precipitate under oxidizing conditions. Sulfide minerals generally are unstable under oxidizing conditions, but may precipitate under reducing conditions. Thus, as a result of the redox-sensitive nature of iron oxides and sulfide minerals, transfer of large amounts of arsenic between these solid phases and neighboring water may result from redox-facilitated precipitation and dissolution reactions (Hinkle and Pollette, 1999).

6.6 Arsenic Concentrations in the Wells G and H Area

6.6.1 Aberjona River Sediment and Surface Water

The Aberjona River watershed area was a center of industrial activity around the beginning of the century. Wastes from tanning, leather finishing, chemical manufacturing (especially sulfuric acid production) and arsenic-based pesticide production were commonly discharged either directly or indirectly into the Aberjona River. The Industri-Plex Superfund Site, located about 1

mile upstream of the Wells G and H area, released an estimated 200 to 900 metric tons of arsenic to the Aberjona watershed (Aurilio et al, 1995).

Studies have shown that arsenic is widespread in the Aberjona River and an estimated 300 metric tons of arsenic is still present in the watershed (Aurilio, 1992). The wetland sediments in the specific area of Wells G and H have 700 kilograms (kg) of arsenic present, The river south of Wells G and H to the Mystic Lakes has an estimated 500 kg of arsenic present. The concentration of arsenic in the Wells G and H area ranges from about 500 to 1000 mg/kg (Knox, 1991). An estimated 100 kg/year of arsenic is transported in surface water towards the Mystic Lakes. The surface water dissolved arsenic concentrations observed by Zeeb (1996) ranged from about 1 to 11 ug/L and the concentration on particulate matter ranged from 200 to 400 mg/kg. Arsenic riverine transport peaked in 1930 at the height of chemical manufacturing and has since declined (Zeeb, 1996). Recent remedial investigations conducted by EPA for the Industri-Plex site in 2001 to 2002 have shown that arsenic concentrations in the Aberjona River have not substantially changed since these previous studies. For example, average concentrations of arsenic in samples collected over an 18-month period under baseflow and storm flow conditions at the outlet of the Halls Brook Holding Area (HBHA), located approximately 0.5 miles upstream of the Wells G and H wetland, were 6.0 ug/L under average dissolved-baseflow and 37.1 ug/L under average total-baseflow conditions and 7.72 ug/L (dissolved) and 29 ug/L (total) under storm flow conditions (TTNUS, 2002). Samples collected during this same period at the outlet of the Wells G and H wetland showed that arsenic concentrations slightly attenuated to an average concentration of 5.6 ug/L during baseflow and a total concentration of 20.1 ug/L during baseflow, indicating that some deposition may be occurring in the Wells G and H wetland (Bullard, 2004).

The calculated arsenic fluxes during the same study period were 6.7 grams per hour (g/hr) during baseflow conditions and 23.78 g/hr during storm flow conditions at the HBHA outlet and 5.3 g/hr during baseflow conditions and 44.8 g/hr during storm flow conditions at the outlet of the Wells G and H wetland. Based on these flux values, it is estimated that the total mass of arsenic that is transported beyond the Wells G and H wetland is between 46 and 78 kilograms per year (kg/year), depending on the number and duration of storm events (Bullard, 2004).

Zeeb (1996) determined that peat layers in proximity to Well H to a depth of 55 centimeters (cm) represent the period of industrialization (past 50 years). These sediments have high levels of arsenic associated with overbank flow across the historical peat surface. An additional 10 to 20 cm of peat may have been impacted through particulate transport by percolation through the open structure of the peat surface. The redox conditions of the impacted peat is anticipated to vary in a cyclic manner due to periodic air entry and surface recharge. Zeeb (1996) indicates that the presence of such high concentrations of arsenic in the upper 80 centimeters of peat over a 50 year period suggests that arsenic transport away from this zone is not significant. The arsenic initially deposited in the wetland was in the form of As(V), most likely sorbed to iron and other hydroxides or precipitated as ferric hydroxy arsenate, Fe_3AsO_4 (Zeeb, 1996). Once buried in saturated anerobic conditions, the reduction of dissolved As(V) to As(III) occurs in a oxidation reduction potential (Eh) range of 70 to 35 millivolts (mV) in the pH range from 6.5 to 7. Amorphous ferric hydroxide dissolves under Eh conditions ranging from 90 to 0 mV. With significant quantities of desorbed As(V) present, ferrous arsenate, $\text{Fe}_3(\text{AsO}_4)_2$, could precipitate

and exist in a narrow Eh-pH zone as ferric iron is reduced to Fe(II) (Zeeb, 1996). Under anaerobic conditions the sulfate present reduces to sulfide and the sulfide reacts with any available iron to produce iron sulfide solids (Smedley and Kinniburgh, 2004). The basic factors favoring iron sulfide solid formation are a sufficient supply of sulfate, iron, high organic matter, presence of sulfate reducing bacteria and an anaerobic environment alternated with limited aeration (Hechanova, 1983).

Data from the Aberjona River Study (M&E, 2004) suggest that sufficient iron and sulfide are present in the peat to support iron-arsenic-sulfur related reactions. The following table summarizes arsenic, iron, and sulfide concentrations in the peat underlying the river and wetlands between Route 128 to the north and Salem Street to the south.

Arsenic, Iron, and Sulfide in Peat – Wells G and H (mg/kg)			
	Arsenic	Iron	Sulfide
Minimum	3.7	2,310	1.605
Maximum	4,550	258,000	121,659
Median	105.5	19,200	558.5

Source: Derived from data presented in M&E, 2004

During the formation of iron sulfide minerals arsenic will be coprecipitated. If insufficient iron is present then thioarsenites will predominate (Wilkin et al., 2003). Moore et al (1988) notes that metal and arsenic sulfide minerals have the potential to sequester a large amount of trace elements, which are potential sources of secondary contamination if exposed to oxidizing environments. Without sulfate limitation, Moore et al found that the main controls on elemental distribution are the precipitation of diagenetic sulfides of arsenic and other metals (e.g., copper and zinc). However, Moore et al do not specify parameters/thresholds for sulfate limited environments. If the sulfides are oxidized they will release the arsenic, which can then be adsorbed to iron oxides thus repeating the cycle. The Industri-Plex facility just upstream of the Wells G and H area was a major producer of sulfuric acid between 1880 and 1929 (Aurilio, 1992) and it is likely that a significant quantity of sulfur compounds were released to the watershed. The sulfide wastes would have been released to the environment through disposal of pyrite cinders, flue dusts and spilled products (sulfuric acid). There are some data available on the levels of sulfides in the Aberjona sediments or regarding the presence of sulfide minerals. Peat sediments in the Aberjona watershed contained between 0.2 to 11-percent by weight of sulfur with a range of 0.2 to 1.6 percent near Wells G and H (Ford, 2004a), so a significant sulfur source is available for the formation of iron sulfides.

Zeeb (1996) identified some arsenic contamination at a deeper peat zone 120 to 160 centimeters below ground surface (bgs) and estimated that about 1 kg of arsenic per meter of river reach was present. Zeeb (1996) noted that this contamination likely migrated deeper through preferential downward groundwater flow possibly due to pumping at Wells G and H under a mobile geochemical window that allowed the arsenic to migrate. Zeeb (1996) assessed whether the peat sediments could be a major source of groundwater contamination to pumped wells and determined that “contrary to previous conclusions, as well as assumptions made as part of several watershed-scale models, the peat soils in this wetland are not loose and permeable, although they are in places interbedded with sand.” Due to the low vertical permeability (6E-5

to 1E-4 centimeters per second [cm/sec]) of the peat, Zeeb (1996) indicates that only a small advective flux would occur through the peat and that contaminant transport through the upper peat would be limited because the peat would be dewatered under pumping conditions. Zeeb (1996) indicates that likely potential sources of arsenic contamination would be outside his study area (which was located between Well H and the river channel). Zeeb postulated that the river channel could be a source, if peat layers are thin or absent.

Ford (2004b) conducted short term, 4 hour, leaching tests of Wells G and H area sediments with aerobic test water. Arsenic leached from the cores at concentrations ranging from 21 to 369 ug/l. In all cases the leached water was at a lower concentration than the initial pore water removed from the sediments. In some cases, the leached concentrations could be explained by residual pore water concentrations, but in other cases over 90-percent of the concentration was due to leaching from the sediment. Ford (2005b) suggests that the degree to which arsenic was leached from these sediments may be positively correlated with Total Organic Carbon (TOC), but four sediment samples may not be sufficient to indicate this with certainty. The leach tests were run on whole sediment samples, including plant fragments. The grain size distribution was not determined for these samples. Ford (2004b) notes that in the field the ability of arsenic to migrate is tied to the hydrologic and geochemistry of the in-situ conditions. The results show that under free draining conditions arsenic can be mobilized. However, as previously noted the mildly reducing weakly oxidized ORP conditions in the Central Area aquifer and the significant presence of iron oxyhydroxide solids is expected to attenuate the mobility of arsenic. In addition, the influx of well oxygenated surface water under pumping conditions is expected to shift the aqueous geochemical equilibrium of arsenic and iron equilibrium to less mobile phases and/or dilute contaminants leached from the peat deposits.

6.6.2 Wells G and H Geochemistry

Dufresne-Henry(1978) indicated that Wells G and H had high levels of dissolved manganese, occasional hydrogen sulfide odors and low oxygen levels, which indicate that reducing conditions were present. The hydrogen sulfide odor is the strongest evidence of reducing conditions. Dufresne-Henry (1978) further indicated that oxygen levels tended to decrease with depth within the potable wells, but did not provide supporting analytical/field data. However, dissolved manganese concentrations can still occur in a mildly oxidizing environment because the manganese redox couple occurs at a higher ORP as compared with iron and arsenic (see Figures 3 and 4). In addition, the long term stability of low iron concentrations from Well G is indicative of mildly oxidizing conditions.

Some observation wells installed by Dufresne-Henry in the wetlands adjacent to Well H had elevated levels of iron but only on two consecutive days during the summer of 1977. For example, Dufresne-Henry observation Well "A", located about 30 feet from Well H, had approximately 10 mg/L of iron at 25 and 50 feet bgs and greater than 40 mg/L at depths of 75 and 92 feet bgs. These data tend to support that reducing conditions were present in some portions of the aquifer during the municipal pumping of Wells G and H. Due to the proximity of Well "A" to Well H, the total iron in the soil is probably similar. However, significantly higher concentrations were detected in groundwater in the wetland. Interestingly, municipal Wells G and H did not have high levels of iron, which suggests that the water quality in the swamp was

entirely different than encountered at Wells G and H. Dufresne-Henry (1978) drew a similar conclusion regarding swamp water quality.

The difference in observed geochemistry in Wells G and H versus monitoring wells installed in, or impacted by, wetland deposits, could be explained in part, by the induced infiltration of well oxygenated and highly oxidized river water. Concentrations of iron in water withdrawn by wells G and H could be diluted by the influx of relatively lower concentration river water. In addition, the relatively oxygenated river water could shift the aqueous geochemical equilibrium predominance of iron to a less soluble (iron (III) oxyhydroxide) form. The influx of river water would have little impact on manganese levels. As shown in the Eh-pH diagram for manganese (Figure 4), the predominance field for manganese is very large and significant precipitation of manganese would not occur except at extremely high pH (pH >8 to 9) for natural water systems or unrealistically high oxidation levels.

6.6.3 General Wells G and H Aquifer Geochemistry

TRC compiled select groundwater inorganic and water quality data from the Central Area aquifer for the period of 1987 to 2002 to prepare composite groundwater concentration isopleths for arsenic, chromium, lead, iron, manganese, dissolved oxygen, ORP, and sulfate. Two sets of concentration isopleths were prepared for arsenic, chromium, lead, iron and manganese. One set is based on maximum groundwater concentration values from low-flow and conventionally collected groundwater samples to provide maximum areal coverage. The second set is based only on low-flow groundwater samples, which provide less areal coverage, but may be more representative since the data are more recent (2001 and 2002) and collected consistent with low-flow protocols. One set of plots were prepared for dissolved oxygen, ORP and sulfate.

The following summarizes the groundwater analytical data sources obtained by PRPs and EPA used to prepare the concentration isopleths. Information concerning these data (source, coverage, data range, limitations, etc.) are summarized in the data matrix presented in Table 3.

- Central Area (OU-2) groundwater data documented in GeoTrans (1994);
- Southwest Properties groundwater data documented in RETEC (1994 and 2003);
- Murphy Waste Oil property groundwater data documented in Clean Harbors (1996 and 1998);
- 2002 groundwater data from the EPA investigation of the Former Drum Disposal Area (FDDA) at the Olympia Property documented in TRC (2002a);
- Central Area groundwater data from eight wells sampled for EPA in October 2002 and documented in TRC (2002c); and
- Split sample data associated with EPA oversight of the RETEC 2002 Supplemental Remedial Investigation of the Southwest Property documented in TRC (2004a).

Table 3. Data Matrix

Data Source	PRP/EPA/Other	Reference Citation ⁽¹⁾	Media Covered by Data Source	Location/Coverage	Data/Date Range	Monitoring Well Depth Ranges	Groundwater Sample Collection Methods	Analytical Methods	Summary of Limitations
Wells G&H Site, Central Area Remedial Investigation Phase IA Report	PRP	GeoTrans, 1994	Groundwater	Central Area	1991-1993 (monitoring wells) 1963/64 – 1991 (Wells G&H)	2.5 – 940 ft bgs	Mix of low flow and pre-low flow protocol collection techniques.	VOCs PAHs Limited Metals (As, Cr, Pb, Na) Limited inorganics (chloride, nitrate, sulfate)	<ul style="list-style-type: none"> Limited heavy metals data for Wells G&H (As, Cr, Pb). Not all samples collected using low flow protocols. Some results potentially biased high due to sampling technique/excess turbidity. No metals data for Source Area (OU-1) properties. Groundwater sampling records not included with the report. Mix of validated and unvalidated data.
Draft Remedial Investigation, Southwest Properties	PRP	RETEC, 1994	Soil, wetland soil, sediments, groundwater, and surface water	Southwest Properties/ Central Area	1993	2.5 - 138 ft bgs	<ul style="list-style-type: none"> Wells purged and sampled via submersible centrifugal pump at rates higher than recommended in low stress/low flow protocols. 	VOCs SVOCs Metals/cyanide Pest/PCBs Various wet chemistry procedures (e.g., nitrates)	<ul style="list-style-type: none"> Documentation does not include groundwater sampling forms (cannot verify turbidity levels or sampling technique). Filtered groundwater inorganics data only. Improper well development technique potentially responsible for high turbidity levels in some wells.
Supplemental Remedial Investigation Report, Southwest Properties	PRP	RETEC, 2003	Soil, wetland soil, sediments, groundwater, and surface water	Southwest Properties/ Central Area	2001 - 2002	3 – 132 ft bgs	<ul style="list-style-type: none"> Used submersible centrifugal pumps to perform low flow sampling. Includes groundwater monitoring data collected by Clean Harbors. 	Various (VOCs, SVOCs, Pest/PCBs, PCB congeners, metals)	Tabulated data not always consistent with validated results.
Corrective Action Investigation Report	PRP	Clean Harbors, 1996	Soil, sediment, wetland soil, groundwater, surface water	Murphy Waste Oil/ Southwest Properties	1994 - 1995	2 – 83 ft bgs	Removed 3 well volumes, sampled using disposable polyethylene bailers	Various (VOCs, SVOCs, Pest/PCBs, metals)	<ul style="list-style-type: none"> Filtered groundwater metals data. Data not validated.
Addendum to Corrective Action Investigation Report (Part II)	PRP	Clean Harbors, 1998	Soil, sediment, wetland soil, groundwater, surface water	Murphy Waste Oil/ Southwest Properties	1998	2.5 – 43.9 ft bgs	Removed 3 well volumes, sampled using either submersible pump or surface mounted peristaltic pump.	VOCs only	<ul style="list-style-type: none"> Filtered groundwater metals data. Data not validated.
Data Summary Report, Wells G&H Superfund Site, Operable Unit 1 – Olympia Property, Woburn, Massachusetts.	EPA	TRC, 2002a	Soil and groundwater	Olympia FDDA and Central Area wetland.	2002	2 – 150	Used bladder pumps to perform low flow sampling (consistent with EPA Region I low stress (low flow) protocols).	VOCs SVOCs Pest/PCBs Metals/cyanide	None noted.

Table 3. Data Matrix

Data Source	PRP/EPA/Other	Reference Citation ⁽¹⁾	Media Covered by Data Source	Location/Coverage	Data/Date Range	Monitoring Well Depth Ranges	Groundwater Sample Collection Methods	Analytical Methods	Summary of Limitations
TRC Central Area Well Data	EPA	TRC, 2002c	Groundwater	Central Area/ Northeast of Well H	2002	17 – 273	<ul style="list-style-type: none"> Used bladder pumps to perform low flow sampling (consistent with EPA Region I low stress (low flow) protocols). 	VOCs SVOCs Pest/PCBs Metals/cyanide	None noted.
EPA/TRC and RETEC Split Report for the Wells G&H – Southwest Properties	EPA	TRC, 2002a	Soil and groundwater	Southwest Properties/ Central Area	2002	3 – 132	<ul style="list-style-type: none"> Used submersible centrifugal pumps to perform low flow sampling. 	Various	None identified.
Wells G&H Source Area Monitoring Data.	PRP	N/A	Groundwater	Source Area Properties	Grace – 1992-2003 UniFirst – 1992-2003 NEP – 1987-2003 Wildwood – 1998-2003	Grace – 10 to 220 UniFirst – 10 to 495 NEP – 12 to 50 Wildwood – 12 to 130 ft bgs	<ul style="list-style-type: none"> Low stress/low flow Passive diffusion samplers (Grace) 	VOCs	Does not include metals/ inorganic data.
Baseline Human Health and Ecological Risk Assessment Report – Wells G&H Superfund Site, Aberjona River Study, Operable Unit 3, Woburn, Massachusetts	EPA	M&E, 2004	Surface water and sediment	Aberjona River	1995-2004	N/A	N/A	VOCs Metals PAHs	None identified.
TTNUS Aberjona River Base and Storm Flow Surface Water Data	EPA	TTNUS, 2002	Surface water	Aberjona River	May 2001 to October 2002	NA	N/A	Metals Water quality parameters (e.g., turbidity)	None identified.
Tighe & Bond Laboratory, Data Report	Other – Collected at end of USGS 30-day pump test by Weston Geophysical Corp.	Tighe & Bond, 1986	Groundwater (pump test effluent)	Municipal wells G and H	1986	Well G – 85 Well H – 88	Not documented.	Metals Wet chemistry Pesticides Radioactivity	<ul style="list-style-type: none"> Not validated. Sampling protocol undocumented.
ETC Laboratory Data Report	Other – Collected 20 days into 30-day USGS pump test by GeoEnvironmental Consultants, Inc.	ETC, 1986	Groundwater (pump test effluent)	Municipal wells G and H	1986	Well G – 85 Well H – 88	Not documented.	VOCs Pest/PCBs Metals Total Plastics	<ul style="list-style-type: none"> Not validated. Undocumented sampling protocol.
Site Assessment Report of the Former Whitney Barrel Company Site	Other – Report prepared for property owner	GHR, 1988	Soil, soil vapor, floor drain sediment, groundwater	Southwest Properties/Whitney Barrel	1987	Approximately 15 feet ⁽²⁾	Pre-low flow/low stress purging protocols.	VOCs SVOCs Pest/PCB Metals	<ul style="list-style-type: none"> Filtered metals analyses. Pre-low stress/low flow sampling protocols.

Notes:

ft bgs – feet below ground surface

(1) – See Section 8 of text for complete reference citation.

(2) – Estimated from available cross-sections – boring/well installation logs not included in the available copy of GHR, 1988.

VOCs – Volatile organic compounds

SVOCs – Semivolatile organic compounds

Pest/PCBs – Pesticides/polychlorinated biphenyls

As – Arsenic

Cr – Chromium

Pb – Lead

FDDA – Former Drum Disposal Area

TOC – Total organic carbon

PAHs – Poly Aromatic Hydrocarbons

TTNUS – TetraTech NUS, Incorporated

There are over 500 monitoring wells in the Wells G and H area that were reviewed to determine water quality characteristics. Due to the layered nature of typical sedimentary strata and associated preferential flow along bedding, the aquifer was divided into 3 separate layers to distinguish between shallow, medium and deep groundwater quality. The groundwater depths to use for each of these designations was determined by plotting all the screen bottom depths for the wells and looking for natural breaks in the frequency of well depths. These breaks occurred at depths of 20 feet and 50 feet as shown in Figure 10.

- Shallow groundwater – 0 to 20 feet below ground surface (bgs)
- Medium Depth Groundwater – greater than 20 to 50 feet bgs
- Deep Groundwater – greater than 50 feet bgs

Concentration plots were prepared for each inorganic parameter at each depth range. Where data is available from multiple rounds of sampling from a particular well, the maximum concentration was plotted. Non-detect results were plotted at the detection limit. Table 4 provides a summary of data statistics for all data in each depth range for the following analytes:

- Arsenic
- Chromium
- Lead
- Iron
- Manganese
- Oxidation-Reduction Conditions
 - Oxidation Reduction Potential (ORP)
 - Dissolved Oxygen (DO)
- Sulfate

The following summarizes the results of the data plotting for each of the analytes by the above-described depth ranges.

Arsenic

Figures 12A through 12F illustrate composite (multiple year) groundwater concentration plots for arsenic. Figures 12A through 12C are based on maximum concentrations using all available data (low-flow and non-low-flow) from each depth range. Figures 12D through 12E use only low-flow data from each depth range collected in 2001 and 2002. Although the low-flow plots provide less areal coverage, they may be more representative of arsenic concentrations, as discussed herein. Each figure depicts the delineated extent of the wetland around municipal Wells G and H. Data used to prepare each plot are summarized in Tables 5A through 5F with well identification, sample date, and concentration in ug/L. The tables also provide the minimum detected concentration, the maximum detected concentration, the average (calculated

with ½ the laboratory reporting limit for non-detect results), and the median value. The median was determined with no adjustment to non-detect values.

Table 4. Data Summary Statistics

Compound	Units	Times Sought	Times Detected	Detection Frequency	Minimum	Maximum	Location of Maximum	MCL/SMCL	Times Exceeded MCL/SMCL
Shallow Wells (0 to 20 feet bgs)									
Arsenic	ug/L	164	90	54.90%	0.55	574	OL-002	10	34
Chromium	ug/L	153	62	40.50%	0.72	1220	OL-002	100	6
Iron	ug/L	67	64	95.50%	25	43700	OL-006	300*	55
Lead	ug/L	159	72	45.30%	0.13	1460	S44	15	25
Manganese	ug/L	69	69	100%	3.5	9010	MW-014S	50*	63
DO	mg/L	51	N/A	N/A	0.00026	27	MW-001S	~	~
ORP	mV	57	N/A	N/A	-134	557	MW-002S	~	~
Sulfate	ug/L	62	62	100%	600	102,000	S93S	250,000*	None
Medium Wells (20 to 50 feet bgs)									
Arsenic	ug/L	134	55	41%	0.13	59.6	BSW-6	10	16
Chromium	ug/L	85	44	51.80%	4	339	S81M	100	6
Iron	ug/L	39	29	74.40%	28.4	48900	BUG1	300*	21
Lead	ug/L	134	59	44%	0.12	120	S22	15	32
Manganese	ug/L	39	39	100%	1.5	39800	S74S	50*	33
DO	mg/L	40	N/A	N/A	0.0001	59	MW-02M	~	~
ORP	mV	52	N/A	N/A	-158	270	OL-2M	~	~
Sulfate	ug/L	42	42	100%	9,000	84,100	IUS2B	250,000*	None
Deep Wells (50 + feet bgs)									
Arsenic	ug/L	126	37	29.40%	0.22	60	S72M	10	3
Chromium	ug/L	130	32	24.60%	2.7	10600	UC12	100	6
Iron	ug/L	46	32	69.60%	49.6	13100	BUG1	300*	22
Lead	ug/L	129	54	41.90%	0.13	270	S40	15	14
Manganese	ug/L	46	45	97.80%	13	16800	S85M	50*	42
DO	mg/L	43	N/A	N/A	0.0001	81	MW-02D	~	~
ORP	ug/L	54	N/A	N/A	-196	255	S94D	~	~
Sulfate	mV	26	26	100%	10,700	150,000	S39	250,000*	None

Notes:

MCL - Maximum Contaminant Limit (EPA)

SMCL - Secondary Maximum Contaminant Limit (EPA)

* - Secondary MCL concentration

~ - No MCL/SMCL for comparison

Figure 12A presents the arsenic concentration plot for the 0 to 20 foot depth interval, which is based on data from 118 monitoring wells. Detected concentrations plotted on Figure 12A ranged from 0.55 ug/L (S93S) to 574 ug/L (OL-002). Thirty (30) wells have detected arsenic concentrations greater than the current EPA MCL of 10 ug/L. Eight areas of elevated arsenic concentrations are illustrated on Figure 12A in areas outlined in red.

Four areas appear to be localized single well highs, including monitoring well IUS2C (59 ug/L) located west of the Route 128 off-ramp, well S7 (49 ug/L) located east of W.R. Grace and Cummings Park, well S21 (12 ug/L) near the former location of the Johnson Brothers greenhouses, well W-MW-3S (62 ug/L) located on the Southwest Properties (specifically Whitney Barrel), and well BSSW-6 (20 ug/L) located near the railroad tracks on the west side of the Wildwood Property.

Three broad areas of elevated arsenic groundwater concentrations in shallow groundwater are located near the Olympia trucking terminal property, near the FDDA and Aberjona wetland, and near Wildwood and the Southwest Properties. The highest concentrations of arsenic detected in shallow (0-20 foot bgs) groundwater were found in the FDDA (OL-002; 574 ug/L) and on or near the Olympia trucking terminal (OL-007, 371 ug/L; OL-006, 267 ug/L; OL-004, 258 ug/L). The next highest concentrations was detected in well S77SS (105 ug/L) on the Wildwood/Aberjona Auto parts property boundary. All of these locations are located in the Aberjona River floodplain. As previously discussed, Aberjona River surface water and sediment have been contaminated by arsenic. Also, in 1970 between 200 and 500 5-gallon containers of arsenic trioxide were discovered and removed from the undeveloped portion of the Olympia Property. Also, in 1971 a drum containing arsenic was found on the property near the Wildwood and Olympia Avenue intersection (TTNUS, 2003).

Several wells with elevated concentrations of arsenic in shallow (0-20 foot bgs) groundwater are present in the area of the Mass Rifle Association and West Cummings Park. Monitoring wells S64S (73 ug/L) and S81S (14 ug/L) lie to the north of the Mass Rifle Association building. Wells DP26 (15 ug/L), S63S (10 ug/L), and S21 (12 ug/L) are located on the West Cummings Park property near the former location of the Johnson Brothers greenhouses.

Some of the maximum concentration data discussed above and/or plotted in Figure 12A were obtained from sampling conducted in 1987, 1991, and 1992 before the commonplace use of low stress/low flow sampling protocols and therefore may not be representative. The following table compares shallow groundwater arsenic results from some of these wells with results from low-stress/low flow sampling conducted on behalf of EPA from the same wells in 2002. This comparison suggests that the elevated concentrations shown in the table may be artifacts of sampling technique; however, the impact of natural attenuation of arsenic over 10+ years cannot be dismissed. Nonetheless, as previously discussed, sorption of arsenic species onto aquifer soils, such as iron oxyhydroxides, is one of the controlling mechanisms for the fate of arsenic.

Arsenic Low-Flow vs. Non-Low-Flow Groundwater Results (0 to 20 Feet)		
Wells	1987/1991/1992 Non-Low-Flow Sampling	EPA 2002 Low Flow (Low Stress) Sampling
S63S	10 ug/L	0.78 ug/L
OL-005	24.1 ug/L	9.3 ug/L
OL-002	574 ug/L	3.2 ug/L
OL-004	258J ug/L	1.6 ug/L

Figure 12B presents the arsenic concentration plot for the >20 to 50 foot bgs depth interval, which is based on data from 86 monitoring wells. Detected concentrations plotted on Figure 12B ranged from 0.33 (MW-014M) to 59.6 ug/L, with the maximum concentration detected at monitoring well BSW6 on the Wildwood Property. Sixteen (16) wells have arsenic concentrations greater than or equal to the current EPA arsenic MCL of 10 ug/L. Two small and two broad areas of arsenic concentrations greater than the MCL are illustrated on Figure 12B in areas outlined in red, two of which represent local single well highs.

The single well highs include the maximum detected concentration at monitoring well BSW6 (59.6 ug/L) on the west side Wildwood Property and east of the railroad right-of-way, and monitoring well S91M (50 ug/L) located between municipal Wells G and H. Swamp deposits tend to thicken (20 feet ±) between well S89 (near Well H) and well S93 (near Well G). Well cluster S91 is located in this area of thicker peat deposits, and the proximity of contaminated peat deposits may account for elevated arsenic detections near well S91M.

One broad area extending from the northwest corner of the Wells G and H site to the Mass Rifle Association property has wells in the >20 to 50 foot depth range with concentrations in excess of the current EPA MCL. Examples include BUG1 (11.2 ug/L), S74S (13 ug/L), S73S (17.9 ug/L) north of Olympia Avenue and monitoring well S82 (21.7 ug/L) north of the Mass Rifle Association.

Another broad area extending from West Cummings Park to the area of the NEP property is characterized by wells with arsenic concentrations in excess of the current EPA MCL. Examples include G15D (10 ug/L) on the Grace Property and well S22 (34.2 ug/L) near the former location of the Johnson Brothers Greenhouses (currently West Cummings Park). Monitoring well S67S (16.3 ug/L) and S67M (23.9 ug/L) are also near the former greenhouses and Cummings Park. EPA1 (9.5 ug/L) and EPA2 (18.7 ug/L) on the NEP property had concentrations approaching and exceeding the arsenic MCL, respectively.

Similar to the above-discussed shallow well arsenic results, some of the maximum results from the > 20 to 50 foot bgs depth interval were obtained before the commonplace use of low stress/low flow sampling protocols and thus may not be representative. The following table compares arsenic results for the >20 to 50 foot bgs depth interval from 1987 and 1991 through 1993 with 2002 low stress/low flow data collected on behalf of EPA.

Arsenic Low-Flow vs. Non-Low-Flow Groundwater Results (>20 to 50 Feet)		
Wells	1987/1991 - 1993 Non-Low-Flow Sampling	EPA 2002 Low Flow (Low Stress) Sampling
S82	21.7 ug/L	0.13J ug/L
S73S	17.9 ug/L (1991) 4.3 ug/L (1992)	3 ug/L
S85S	1.4 ug/L	<0.1 ug/L

Figure 12C presents the arsenic concentration plot for wells greater than 50 feet in depth, which is based on data from 65 monitoring wells. Detected concentrations plotted on Figure 12C range from 0.22 ug/L (UC11) to 60 ug/L (S72M). Three (3) wells with concentrations of arsenic greater than the current EPA MCL of 10 ug/L are illustrated in areas outlined in red. These wells include IUS2A (42.6 ug/L) located west of the Route 128 off ramp, S72M (60 ug/L) on the Olympia Terminal property, and S79D (32 ug/L), located west of the railroad right-of-way near the Wildwood property. Figure 11C illustrates the detection of arsenic in Wells G and H from monitoring conducted in September 1979 following the shutdown of the wells. Both detections were less than the current EPA MCL for arsenic.

As with the shallow (>20 feet bgs) and medium depth (>20 to 50 feet bgs) arsenic results, some of the maximum values may be biased high due to non-low flow/low stress sampling techniques. The following table compares arsenic results for the >50 foot bgs depth interval from 1991 through 1993 with 2002 low stress/low flow data collected on behalf of EPA.

Arsenic Low-Flow vs. Non-Low-Flow Groundwater Results (> 50 Feet)		
Wells	1991 - 1993 Non-Low-Flow Sampling	EPA 2002 Low Flow (Low Stress) Sampling
S93D	2.9 ug/L	0.87 ug/L
S72M	60 ug/L (1991) 4.5 ug/L (1993)	4 ug/L
S72D	20 ug/L (1991) 10 ug/L (1991) 2.3 ug/L (1993)	2.6 ug/L

As noted above, some of the maximum values plotted in Figures 12A through 12C may be biased high due to artifacts from non-low-flow sampling. Figures 12D through 12F were prepared using only low-flow data collected in 2001 and 2002. In general, concentrations plotted are lower in concentration, which is attributable to more representative sampling technique (i.e., low-flow). However, an area of significantly elevated arsenic concentration is evident on the Olympia trucking terminal property. Also, the highest concentrations are generally located in shallow groundwater in the Central River Valley.

Chromium

Figures 13A through 13F illustrate composite groundwater concentration plots for chromium. Figures 13A through 13C are based on maximum concentrations using all available groundwater data (low-flow and non-low-flow) from each depth range. Figures 13D through 13F use only low-flow groundwater data from each depth range collected in 2001 and 2002. Although Figures 13D through 13F provide less areal coverage than the A through C isopleth series, they may be more representative of chromium concentrations in the Central Area aquifer. Each figure depicts the delineated extent of the wetland around municipal Wells G and H. Data used to prepare each plot are summarized in Tables 6A through 6F, which includes well identification, sample date, and concentration in ug/L. The tables also provide the minimum detected concentration, the maximum detected concentration, the average concentration (calculated with ½ the laboratory reporting limit for non-detect results) and the median value. The median was determined with no adjustment to non-detect values.

Figure 13A includes chromium data from 118 monitoring wells in this depth range. Detected concentrations ranged from 0.72 ug/L (MR-MW-1) to 1,220 ug/L (OL-002). Seven (7) wells have chromium concentrations greater than or equal to the EPA MCL of 100 ug/L for total chromium.

Two general areas with concentrations greater than the MCL are shown outlined in red in Figure 13A. One large and small areas of elevated chromium concentrations outlined in red covering an

area from the Olympia trucking terminal property (OL-1, 281 ug/L; OL-4, 399 ug/L) to the FDDA (OL-2, 1220 ug/L).

The second area includes portions of the Mass Rifle Association property (S64S; 326 ug/L) and the UniFirst property (S71S; 162 ug/L). However, these maximum data were obtained from sampling conducted in 1987 and 1991, before the commonplace use of low stress/low flow protocols and therefore may not be representative. The following table compares shallow groundwater chromium results from some of these wells with results from sampling conducted on behalf of EPA from the same wells in 2002 using low flow (low stress) sampling protocols. This comparison suggests that the elevated concentrations illustrated on the figure may be artifacts of sampling technique. As previously discussed, aqueous partitioning coefficient (Kd) values for Cr (III) are consistent with strong adsorption tendencies with aquifer solids. Consequently, the 1987/1991 results may be biased high by excessive solids entrainment during well purging and sampling. Groundwater sampling records were not included with some of the data sources (i.e., GeoTrans, 1994).

Chromium Low-Flow vs. Non-Low-Flow Groundwater Results (0 to 20 Feet)		
Wells	1987/1991 Non-Low-Flow Sampling	EPA 2002 Low Flow (Low Stress) Sampling
OL-1	281 ug/L	<2.5 ug/L
OL-2	1220 ug/L	<2.5 ug/L
OL-3	19 ug/L	<2.5 ug/L
OL-4	399 ug/L	<2.5 ug/L
S63S	100 ug/L	<2.5 ug/L

Figure 13B presents the chromium concentration plot for the >20 to 50 foot depth interval, which is based on data from 84 monitoring wells. Detected concentrations range from 3.1 ug/L (DP7) to 339 ug/L (S81M). Six (6) wells have chromium concentrations greater than the EPA MCL of 100 ug/L, including S67M (102 ug/L), S64M (106 ug/L), S22 (148 ug/L), S67S (172 ug/L), S65M (242 ug/L), and the maximum at S81M (339 ug/L). Three areas of elevated chromium concentrations are outlined on Figure 13B in red, which are associated with the MCL exceedances. These include the maximum detected concentration in this depth range at monitoring well S81M (339 ug/L) on the Mass Rifle Association property west of the Olympia trucking terminal, wells S67S (172 ug/L) and S67M (102 ug/L) in Cummings Park near the former locations of the Johnson Brothers greenhouses, and well S65M (242 ug/L) at the rear of West Cummings Park and east of the Mass Rifle Association property. All of the identified MCL exceedances are from 1985 and 1991 sampling events and none were identified as collected by low flow protocols; therefore, these results may be biased high due to excessive solids entrainment. Groundwater sampling records were not included with some of the data sources; therefore, this hypothesis cannot be further investigated at this time.

Figure 13C presents the chromium concentration plot for wells greater than 50 feet in depth, which is based on data from 65 monitoring wells. Detected concentrations range from 2.7 ug/L (BUG1) to 10,600 ug/L (UC-12). Five (5) wells have chromium concentrations greater than the EPA MCL of 100 ug/L, including UC11 (131 ug/L), MR-MW-5D (133 ug/L), UC13 (304 ug/L), UC14 (9,040 ug/L), and UC12 (10,600 ug/L). UC11 through UC13 are located on Cummings

Park/West Cummings Park Property in the vicinity of the Former Johnson Brothers Greenhouse. Well UC14 is located southeast of the Charette building. All of the identified MCL exceedances are from 1985 and 1991 sampling events and none were identified as collected by low flow protocols; therefore these results may be biased high due to excessive solids entrainment. Wells UC12 (10,600 ug/L), and UC14 (9,040 ug/L) are the two highest chromium results in the database.

MR-MW-5D (133 ug/L) is located on the Southwest Properties, specifically the Murphy Waste Oil property near the Murphy Wetland. The Murphy Wetland is an area of elevated chromium concentrations in sediment; the chromium III-contamination in the sediments in the Murphy Wetland was identified as a risk driver in the Baseline Ecological Risk Assessment for the Southwest Properties (TRC, 2004). The data plotted for MW-5D from 11/06/2001 was collected using low stress/low flow sampling protocols and is considered representative (i.e., not impacted by excessive solids entrainment).

As discussed above, some of the maximum chromium concentration values plotted in Figures 13A through 13C may be biased high due to artifacts from non-low-flow sampling. Figures 13D through 13F were prepared using only low-flow groundwater data collected in 2001 and 2002. In general, concentrations plotted are lower in concentration, which is attributable to more representative sampling technique (i.e., low-flow). In addition, only one low-flow result exceeded the total chromium MCL of 100 ug/L (MR-MW-5D, 133 ug/L, Figure 13F).

Lead

Figures 14A through 14F illustrate composite (multiple year) groundwater concentration plots for lead. Figures 14A through 14C are based on maximum concentrations using all available data (low-flow and non-low-flow) from each depth range. Figures 14D through 14F use only low-flow groundwater data from each depth range collected in 2001 and 2002. Figures 14D through 14F are based on low-flow groundwater data only. Although Figures 14D through 14F provide less areal coverage than the A through C series, they may be more representative of lead concentrations in the Central Area aquifer. Each figure also depicts the delineated extent of the wetland around municipal Wells G and H. Data used to prepare each plot are summarized in Tables 7A through 7F, which includes well identification, sample date, and concentration in ug/L. The tables also provide the minimum detected concentration, the maximum detected concentration, the average concentration (calculated with $\frac{1}{2}$ the laboratory report limit for non-detect results), and median value. The median was determined with no adjustment to non-detect values.

Figure 14A presents the lead concentration plot for the shallow (0 to 20 foot) depth interval, which includes lead data from 111 monitoring wells in this depth range. Detected concentrations ranged from 0.13 ug/L (MW-001S) to 1,460 ug/L (S44). Twenty-two (22) wells have lead concentrations greater than or equal to the EPA Safe Drinking Water Act (SDWA) Action Level of 15 ug/L. Several areas with concentrations greater than the lead Action Level are shown outlined in red in Figure 14A. The largest area extends from the Olympia trucking terminal, through the Mass Rifle Association, to the UniFirst property, areas of Cummings Park and near to the former Independent Tallow property, and includes wells OL-1 (79.6 ug/L), OL-4 (192

ug/L), S81S (51 ug/L), S64S (168 ug/L), S65S (115 ug/L), DP6S (69.1 ug/L), S21 (83 ug/L), UC5 (51.4 ug/L), IUS2C (25.2 ug/L), IUS3C (35.7 ug/L), and S44 (1,460 ug/L) near the former Independent Tallow property. Lead contaminated groundwater was also identified at the FDDA on the Olympia Property, with a concentration of 188 ug/L at OL-2, and on the Wildwood Property including wells S78S (96 ug/L) and BSSW6 (71 ug/L). Single well highs are located on the Southwest Properties (MR-2SS; 29 ug/L), Cummings Park near the Routes 93 and 128 interchange (S7; 520 ug/L), and S91S (19.7 ug/L) between Wells G and H.

However, many the results in excess of the lead Action Level were obtained from sampling conducted in 1987, 1991, 1992, and 1993, before the commonplace use of low stress/low flow protocols and therefore may not be representative due to excessive entrainment of solids. The following table compares shallow groundwater (0-20 foot bgs) lead results from some of these wells with results from 2002 sampling conducted on behalf of EPA from the same wells using the Region I low stress (low flow) protocol. This comparison suggests that some of the elevated concentrations illustrated on Figure 14A from 1987-1993 sampling events may be artifacts of sampling technique. As previously discussed, the Langmuir sorption constants for lead are consistent with strong adsorption tendencies with aquifer solids. Consequently, the 1987/1991-1993 results may be biased high by excessive solids entrainment during well purging and sampling. Groundwater sampling records were not included with some of the data sources (i.e., GeoTrans, 1994).

Lead Low-Flow vs. Non-Low-Flow Groundwater Results (0 to 20 Feet)		
Wells	1987/1991-1993 Non-Low-Flow Sampling	EPA 2002 Low Stress (Low Flow) Sampling
OL-1	79.6 ug/L	<0.1 ug/L
OL-2	188 ug/L	<0.1 ug/L
OL-4	192 ug/L	0.43 ug/L
S63S	62 J ug/L	0.97 ug/L

Figure 14B presents the lead concentration plot for the >20 to 50 foot depth interval, which is based on data from 85 monitoring wells. Detected concentrations range from 0.15 ug/L (S92M) to 120 ug/L (S22). Nineteen (19) wells have lead concentrations greater than the EPA SDWA Action Level of 15 ug/L. The plotted data suggests widespread contamination in excess of the Action Level, with isolated wells with results below the action level. The maximum lead concentration from this depth interval is 120 ug/L at S22 located in West Cummings Park near the former location of the Johnson Brothers greenhouses. The second highest lead concentration detected in this depth interval is at S91M (54.9 ug/L), with is located between wells G and H. All of the results in excess of the Action Level were from samples conducted in the 1981 to 1993 time frame before the commonplace use of low stress (low flow) sampling protocols and therefore may not be representative. The following table compares >20 to 50 foot bgs groundwater lead results with available 2002 sampling conducted on behalf of EPA from the same wells using Region I low flow (low stress) protocols. This comparison suggests that some of the elevated concentrations illustrated on the figure maybe artifacts of sampling technique (i.e., excessive solids entrainment). As previously discussed, the Langmuir sorption constants

for lead are consistent with strong adsorption tendencies with aquifer solids. Consequently, the 1981 to 1993 results may be biased high by excessive solids entrainment during well purging and sampling. Groundwater sampling records were not included with some of the data sources (i.e., GeoTrans, 1994).

Lead Low-Flow vs. Non-Low-Flow Groundwater Results (> 20 to 50 Feet)		
Wells	1981 to 1993 Non-Low-Flow Sampling	EPA 2002 Low Stress (Low Flow) Sampling
S63D	32.8 ug/L	0.25 ug/L
S72S	22 ug/L	<0.1 ug/L
S82	35 ug/L	0.12J ug/L
S85S	9 ug/L	< 0.1 ug/L

Figure 14C presents the lead concentration plot for the > 50 foot depth interval, which is based on data from 68 monitoring wells. Detected concentrations range from 0.13 ug/L (GEO-1) to 270 ug/L (S40 or Well G), with the maximum concentration detected in well S40 (also known as Well G). Eight (8) wells have lead concentrations greater than or equal to the EPA SDWA Action Level of 15 ug/L. These wells include UG4 (15 ug/L), S93D (15 ug/L), NEP2 (17 ug/L), IUS2A (20.5 ug/L), UC14 (48.9 ug/L), S90D (81 ug/L), UC13 (124 ug/L), and S40 (a.k.a. Well G) (270 ug/L). All of the results equal to or in excess of the Action Level were from samples collected in the 1981 to 1991 time frame before the commonplace use of low stress/low flow protocols and therefore may not be representative. The following table compares the greater than 50-foot depth groundwater lead results with available 2002 sampling conducted on behalf of EPA from the same wells using low stress/low flow protocols. This comparison also suggests that some of the elevated concentrations illustrated on the figure maybe artifacts of sampling technique. As previously discussed, the Langmuir sorption constants for lead are consistent with strong adsorption tendencies with aquifer solids. Consequently, the 1981 to 1991 results may be biased high by excessive solids entrainment during well purging and sampling. Groundwater sampling records were not included with some of the data sources (i.e., GeoTrans, 1994).

Lead Low-Flow vs. Non-Low-Flow Groundwater Results (> 50 Feet)		
Wells	1981 to 1991 Non-Low-Flow Sampling	EPA 2002 Low Stress (Low Flow) Sampling
S72M	6.31 ug/L	<0.1 ug/L
S72D	6 ug/L	0.33 ug/L
S91D	13 ug/L	0.51 ug/L
S93D	15 ug/L	0.43 ug/L

The detection of lead in Well G (S40) was anomalous. The sample was collected on 1/25/81. No prior or subsequent monitoring detected lead in Wells G or H except at the end of the USGS aquifer pump test on 1/3/1986. Lead was detected in aquifer pump test effluent from Wells G and H at concentrations of 90 ug/L and 80 ug/L, respectively. However, samples collected 10 days prior during the aquifer pump test did not detect lead at a laboratory reporting limit of 50

ug/L. Later sampling unrelated to the pump test conducted on 8/21/91 did not detect lead at a laboratory reporting limit of 5 ug/L. The erratic detections are contrary to the limited mobility of lead in groundwater and could be artifacts of sampling technique (none of the historical data were based on low-flow sample collection) or could reflect another source of contamination. For example, brass or bronze parts on the pump could serve as a source of detectable lead (WWJ, 1994). The following table summarizes available lead analytical data from groundwater samples from Wells G (S40) and H (S39).

Summary of Available Lead Groundwater Data from Wells G and H (ug/L)			
Date	Well G (S40)	Well H (S39)	Reference
9/24/79	"0"	"0"	1
1/25/81	270	<40	1
12/23/85 (20 days into the USGS pump test)	<50	<50	2
12/24/85	<5	<5	1
1/3/86 (at conclusion of 30 day USGS pump test)	90	80	3
8/21/91	<5	<5	1

Notes

- 1 – GeoTrans, 1994
- 2 – ETC, 1986
- 3 – Tighe & Bond, 1986

As noted above, some of the maximum values plotted in Figures 14A through 14C may be biased high due to artifacts from low-flow sampling. Figures 14D through 14F were prepared using only low-flow data collected in 2001 and 2002. The low-flow data are generally lower in concentration, which is attributable to more representative sampling technique (i.e., low-flow).

Only four low-flow results exceeded the lead action level of 15 ug/L (MR-MW-14, 62.7 ug/L; MR-MW-16, 42.5 ug/L; and MR-MW-21, 15.5 ug/L on Figure 14D; and S83, 20 ug/L on Figure 14F). All four wells with low-flow concentrations of lead in excess of the action level are located at the Southwest Properties.

Iron

Figures 15A through 15F illustrate composite (multiple year) groundwater concentration plots for iron. Figures 15A through 15C were prepared using maximum concentrations from each well using all available data (low-flow and non-low-flow) for each depth range. Figures 15D through 15F use only the available low-flow groundwater from 2001 and 2002, which may be more representative of iron concentration in the Central Area aquifer. However the low-flow figure series provides less areal coverage. All iron plots depict the delineated extent of the wetland around municipal Wells G and H. Data used to prepare each plot are summarized in Tables 8A through 8F, which includes well identification, sample date, and concentration in ug/L. The tables also provide the minimum detected concentration, the maximum detected concentration, the average concentration (calculated with ½ the laboratory reporting limit for

non-detect results), and the median value. The median was determined with no adjustment to non-detect values.

Figure 15A presents the iron concentration plot for the shallow (0 to 20 foot) depth interval, which is based on iron data from 54 monitoring wells in this depth range. Detected concentrations plotted in Figure 15A ranged from 39.9 ug/L (MR-MW-2) to 43,700 ug/L (OL-006). Forty-six (46) wells in this depth interval have iron concentrations greater than or equal to the EPA Secondary MCL (SMCL) of 300 ug/L. Figure 15A suggest the wide spread presence of iron in excess of the SMCL in the Central Area aquifer. Examples of the highest iron concentrations are found in the following locations:

- Olympia trucking terminal property: OL-6 (43,700 ug/L), OL-7 (25,100 ug/L).
- Wells G and H Wetland: MW-003 (25,000 ug/L), S88S (42,200 ug/L), MW-004 (25,200 ug/L), and MW-007 (21,300 ug/L).
- Southwest Properties: MW-4S (35,000 ug/L), MW-7 (21,300 ug/L), WB-1SS (20,000 ug/L), MW-3 (25,000 ug/L), and AB-1 (19,100 ug/L).

Figure 15B presents the iron concentration plot for the >20 to 50 foot depth interval, which is based on data from 29 monitoring wells. Detected concentrations range from 28.4 ug/L (S66D) to 48,900 ug/L (BUG-1). Eighteen (18) wells have iron concentrations greater than the EPA SMCL of 300 ug/L. Figure 15B also suggests the wide spread presence of iron in excess of the SMCL in the Central Area aquifer. The highest concentrations tended to be located near the Aberjona River or the wetland. Examples include BUG1 (48,900 ug/L), S74S (22,300 ug/L), S72S (15,500 ug/L), S91M (18,400 ug/L), and S93M (19,300 ug/L).

Figure 15C presents the iron concentration plot for wells greater than 50 feet in depth, which is based on data from 31 monitoring wells. Detected concentrations range from 49.6 ug/L (S88D) to 13,100 ug/L (BUG-1). Sixteen (16) wells in this depth range have iron concentrations greater than the EPA SMCL of 300 ug/L. Figure 15C also suggests the widespread presence of iron in excess of the SMCL in the Central Area aquifer. The highest concentrations also tended to be located near the Aberjona River or the wetland. Examples include BUG1 (13,100 ug/L), S72M (12,500 ug/L), S72D (8,330 ug/L), and S93D (4,220 ug/L)

Most of the plotted iron data are from sampling conducted for, or overseen by, EPA in 2002 and were collected consistent with the EPA Region 1 low stress (low flow) protocol. Consequently, analytical bias associated with excessive entrainment of particulates associated with improper sampling technique does not impact most of the above-discussed iron SMCL exceedances. Figures 15D through 15F were prepared using only the available low-flow sampling data from 2001 and 2002, and illustrate a similar pattern of generally elevated iron concentrations (i.e., <SMCL) in the Central Area aquifer. This may reflect the limited amount of non-low-flow iron data in the database.

Manganese

Figures 16A through 16F illustrate composite (multiple year) groundwater concentration isopleths for manganese. The manganese concentration plots in Figures 16A through 16C are

based on maximum groundwater concentrations using all available data (low-flow and non-low-flow) for each depth range. Figures 16D through 16F are a series of manganese concentration plots prepared using only low-flow groundwater data from 2001 and 2002 for the three depth intervals. The low-flow series of figures provide less areal coverage, but may be more representative of manganese concentrations in the Central Area aquifer. Maximum data used to prepare each plot are summarized in Tables 9A through 9F. The tables also provide the minimum detected concentration, the maximum detected concentration, the average concentration (calculated with $\frac{1}{2}$ the laboratory reporting limit for non-detect results), and the median value. The median was determined with no adjustment to non-detect values.

Figure 16A presents the manganese concentration plot for the shallow (0 to 20 foot) depth interval, which includes manganese data from 53 monitoring wells in this depth range. Manganese was detected in all wells in this depth range. Concentrations ranged from 3.5 ug/L (MR-MW-2) to 9,010 ug/L (MW-14S). All but two wells had manganese concentrations in excess of the EPA SMCL of 50 ug/L. These data document the wide spread presence of manganese in excess of the SMCL in the Central Area aquifer.

Figure 16B presents the manganese concentration plot for the intermediate depth range (>20 feet to 50 feet bgs), which includes data from 29 monitoring wells in this depth range. Manganese was detected in all wells in this depth range. Concentrations ranged from 2.8 ug/L (S92M) to 39,800 ug/L (S74S). All but four wells had manganese concentrations in excess of the EPA SMCL of 50 ug/L. These data also document the wide-spread presence of manganese in excess of the SMCL in the Central Area aquifer.

Figure 16C presents the manganese concentration plot for wells greater than 50 feet in depth, which is based on data from 31 monitoring wells in this depth range. Concentrations ranged from 18.2 ug/L (S92D) to 16,800 ug/L (S85M). All but one well (S92D; 18.2 mg/L) had results in excess of the manganese SMCL.

Most of the data depicted in Figures 16A through 16C are from sampling conducted for, or overseen by, EPA in 2002 and were collected consistent with the EPA Region 1 low stress/low flow protocol. Consequently, analytical bias associated with excessive entrainment of particulates associated with improper sampling technique does not impact most of the noted SMCL exceedances. Figures 16D through 16F were prepared using only the available low-flow sampling data for manganese from 2001 and 2002. The low-flow only plots also document a pattern of elevated manganese concentrations (i.e., >SMCL) in the Central Area aquifer. Only 10 out of the 97 wells plotted in the low-flow (Figure 16B through 16F) are less than the EPA manganese SMCL of 50 ug/L.

Oxidation Reduction Potential and REDOX Conditions

Figures 17A, 17B, and 17C illustrate ORP readings in the three defined depth intervals in the aquifer. Data used to prepare each plot are summarized in Tables 10A through 10C for each depth interval. The tables include well identification, sample date, and ORP values in mV. The Tables also provide the minimum and maximum ORP values, the average ORP value and the median value.

ORP values in Central Area aquifer groundwater are variable. ORP values from Central Area monitoring wells tend to exhibit at most moderately reducing conditions (-94 mV to -168 mg mV), with other areas exhibiting oxidizing conditions ranging as high as 270 to 556 mV. Areas exhibiting oxidizing conditions tend to be located in upland areas approaching the sides of the buried bedrock valley. The sides of the buried bedrock valley and flood plain tend to have downward vertical groundwater gradients, and therefore these areas may reflect the influx of fresh, oxidized meteoric water. Groundwater in the center of the valley and flood plain, near where wells G and H are located as well as the extensive wetland/peat deposits, tends to exhibit moderately reducing conditions. There are exceptions to this generalized model that may reflect localized recharge of fresh meteoric or oxidizing river water, potential influx of oxidizing water at depth from local preferential pathways for fresh meteoric water (e.g., bedrock fractures), or potential measurement error. In general, the lowest ORP values measured are only moderately reducing.

Dissolved oxygen (DO) measurements approximately mirror the ORP values as illustrated in Figures 18A, 18B, and 18C. Data used to prepare each plot are summarized in Tables 11A through 11C. The tables include well identification, sample date, and the DO reading. The tables also provide the minimum and maximum DO readings, the average DO reading, and the median value. As expected, low DO concentrations are evident in areas with low ORP values, and higher DO values are observed in areas with higher ORP values.

The percentage of ferrous iron in groundwater is more indicative of general redox conditions since the redox break point between ferric iron (Fe[III]) and ferrous iron (Fe[II]) is similar to that of As(V) and As(III). In the sequence of redox change from oxidizing to reducing conditions, As(V) reduction would normally be expected to occur after Fe(III) reduction, but before sulfate (SO_4^{2-}) reduction (Smedley and Kinniburgh, 2004).

Recent research highlights the importance of ORP to arsenic transformation, suggesting that oxidation of As(III) to As(V) can occur in relatively reducing conditions and near neutral pH. Given the moderately reducing to oxidizing conditions evident in the Central Area aquifer and the mildly acidic/near neutral pH range, conditions favoring less mobile As(V) may be present in site groundwater, which might explain the very low concentration to non-detect arsenic results from Wells G and H.

Note that under pumping conditions, the induced influx of well oxygenated/highly oxidized river water is expected to further shift aqueous redox equilibria the aquifer toward less reducing/more oxidizing conditions, especially along flow paths. Given the depth of wells G and H, the shift in

equilibrium towards more oxidizing conditions would be expected in the shallow, medium, and deep aquifer under pumping conditions.

Sulfate

Figures 19A, 19B, and 19C illustrate composite (multiple year) groundwater concentration isopleths for sulfate. Maximum sulfate concentration data used to prepare each plot are summarized in Tables 10A through 10C. The tables also provide the minimum detected concentration, the maximum detected concentration, the average concentration, and the median value.

Figure 19A presents the sulfate groundwater concentration plot for the shallow (0 to 20 foot) depth interval, which includes data from 62 monitoring wells in this depth range. Sulfate was detected in all wells in this depth range. Concentrations ranged from 600 ug/L to 102,000 ug/L. None exceeded the EPA SMCL of 250,000 ug/L. These data document the widespread presence of sulfate in the Central Area aquifer.

Figure 19B presents the sulfate groundwater concentration plot for the intermediate depth range (>20 feet to 50 feet bgs), which includes data from 42 monitoring wells in this depth range. Sulfate was detected in all 42 wells. Concentrations ranged from 9,000 ug/L to 84,100 ug/L. None exceeded the EPA SMCL (250,000 ug/L). These data also document the widespread presence of sulfate in the Central Area aquifer.

Figure 19C presents the sulfate groundwater concentration plots for wells greater than 50 feet in depth, which is based on data from 26 monitoring wells in this depth range. Concentrations ranged from 10,700 ug/L to 150,000 ug/L, and further document the widespread presence of sulfate in the Central Area aquifer. None exceeded the EPA SMCL (250,000 ug/L).

Smedley and Kinniburgh (2004) note that sulfate levels less than 1,000 ug/L (1 mg/L) are one of the characteristic features of high-arsenic groundwaters (see Section 6.4). Only one measurement (600 ug/L in Well DP14 in the 0 to 20 foot depth interval) was less than 1,000 ug/L. Under anaerobic conditions, sulfate ion is reduced to sulfide ion, which establishes an equilibrium with hydrogen ion to form hydrogen sulfide (Sawyer and McCarty, 1978). Hence, concentrations less than 1,000 ug/L are suggestive of reducing conditions, which in turn are favorable for the more soluble and mobile As(III) form of arsenic. The high sulfate concentrations observed throughout the Central Area aquifer and at all depth ranges suggest generally less reducing/more oxidized aqueous chemical environment.

6.7 Hydrologic Factors Controlling the Migration of Arsenic

Arsenic is known to be present in the Aberjona River watershed especially near Wells G and H Superfund Site wetlands (peat). The presence of the arsenic poses a potential threat to potable water development, but whether it will pose an actual threat depends on the geochemical and hydrologic properties of the aquifer. If the arsenic is mobile, arsenic contaminated groundwater might not get to the wells in sufficient quantity to exceed drinking water standards. The key to

this analysis is the hydraulic properties of the peat sediments, which contain most of the arsenic contamination in the Wells G and H wetland.

Bialon (1995) conducted testing of hydraulic properties for 22 samples of various types of peat including live root mat, Typha peat, Sedge peat and Red Woody peat at several locations. Bialon (1995) determined that the variation in the hydraulic properties between the wetland layers are very small. The range of in-situ hydraulic conductivity throughout the wetland layers are between $2E-6$ and $6E-4$ cm/sec. For comparison, this hydraulic conductivity range is equivalent to the hydraulic conductivity of a typical silt.

Zeeb (1996) developed a groundwater model for the Wells G and H area using representative hydraulic conductivities for the peat layer. He determined that the Aberjona River wetland area influenced by drawdown from Well H was a 300 meter reach of the river. He calculated that leakage from the 300 meter reach of the river peat unit is 0.03 cubic feet per second (cfs) compared to the total well H flow rate of 400 gpm or 0.89 cfs. Based on these values the amount of water flowing through the peat to well H is 3.6-percent of the total flow from well H. This ratio shows that a high rate of dilution (96.4-percent) is provided for groundwater originating in the peat layer. Based on USGS (1987), the actual amount of water supplied to Wells G and H from surface water supplies was about 550 gpm. Some of this water may have been derived from induced infiltration from the Aberjona River caused by the excavation of peat sediments near the well field with emplacement of sand and gravel in a radius of 30 feet around the well as described in Dufresne-Henry (1978). The amount of downhole leakage through the gravel packed wells may have been about 20 gpm per well assuming a gravel pack area (in section) of 9.4 square feet, a hydraulic conductivity (K) of 3,000 feet per day (ft/d) and a gradient of 10 feet/70 feet. Alternatively, the effect of the peat removal could be even larger since the placement of the gravel may have opened a 30 foot radius conduit to the well screen. With the peat removed, the implaced gravel would be underlain by the sand and gravel glacial aquifer opening up a highly permeable preferential flow pathway. Other wells that may be placed in the Central Area (OU-2) aquifer could have lower discharge rates or have water derived from significantly different strata, on a proportional basis, if this short circuiting to surface water were not in place. However, other wells appear to have sufficient yield even when the surface connection like that observed by Wells G and H is discounted (e.g., the Riley supply wells). The glacial aquifer consisting of stratified sand and gravel has a high hydraulic conductivity ranging from 130 to 215 feet per day. The glacial aquifer is about 82 feet thick in comparison to the peat layer which is about 1 to 5 feet thick in most places. Given the low hydraulic conductivity and thinness of the peat in comparison to the glacial aquifer, the amount of water that would be supplied by the glacial aquifer is more than one-thousand times higher, especially considering the deep screen depths of Wells G and H.

There are a very limited number of studies showing arsenic retardation rates in groundwater. A study by Throssell and Blesing (2003) indicated that the arsenic flux rate in a sandy aquifer was 8-percent of the rate of un-retarded groundwater.

6.8 Influence of the Peat Deposits and Organic Material

Where organic material is present, it exerts a significant influence on the presence of arsenic and iron. Since reduction of FeOOH is facilitated by microbial mediation, which is driven by metabolism of organic matter, there will be a greater tendency to release arsenic to solution from the dissolution of FeOOH solids where organic material is present (Ravenscroft et al, 2001). Arsenic is also thought to partition with sulfide minerals in anoxic soils and sediments. As(III) can undergo conversion to a sulfur-containing compound that is subsequently sequestered through its reduction to, and subsequent precipitation as, an FeAsS-like phase. Surface reactions of iron sulfide minerals, specifically FeS₂, are responsible for the initial attenuation of arsenic in such environments (Bostick et al, 2004).

The Aberjona River valley near Wells G and H is underlain by extensive peat deposits. Zeeb (1996) encountered peat and diatomaceous silt deposits as thick as 7 meters near Well H, although this may reflect the local presence of a former glacial kettlehole. Peat deposits encountered in the wetland during EPA's 2002 investigation of the Olympia site were approximately 2 feet thick. The areal extent of the peat depositions, and degree to which they may have been incised by the action of the river over time is not known; however, the wetland is approximately 38-acres. As noted previously, within the 0 to 20 feet depth zone most of the wells with arsenic concentrations in excess of the current EPA MCL are located near the river and in the wetland in the center of the Aberjona River valley near municipal wells G and H. Conditions in the peat are reasonably expected to be anaerobic and thus favor the more mobile As(III) form of arsenic. In addition, the presence of the organic peat material should serve as an ample energy source for microbial transformation of FeOOH solids, thus further contributing to the presence of dissolved arsenic. Although none of the monitoring wells in the 0 to 20 range were screened in the peat formation, the small scale circulation of water and solutes between the peat horizon and the underlying sand and gravel aquifer would provide a mechanism for redistributing organic and inorganic solutes in this zone, thus leading to the relative enrichment in dissolved arsenic concentrations in this shallow zone.

Within the glacial sand and gravel aquifer, there are significantly lower concentrations of natural organic material measured as Total Organic Carbon (TOC). TOC measurements conducted in the Central Area aquifer during EPA's 2002 investigation of the Olympia site encountered TOC concentrations ranging from 103J (estimated) to 3,180 mg/kg, which is significantly less than the high percentage TOC content in a peat deposit. Furthermore, the organic carbon encountered at depth in the aquifer was undoubtedly deposited via fluvial processes during the retreat of the glacier and was likely oxidized to varying degrees before deposition. This type of exposure would have degraded the most reactive components of the organic material, leaving it cellulose-rich and unfavorable for bacterial metabolism. The combination of low organic material concentrations and low organic material quality are insufficient to drive redox to the degree evident in or near the shallow peat beds in the aquifer. As a consequence, FeOOH solids are less likely to be reduced at depth and thus remain available to serve as a sorption sink for arsenic, whether As(V) or As(III). (As[III] sorbs less strongly to FeOOH solids compared to As(V), but still can sorb.)

The thickness of the peat deposits is likely not uniform and may be greatest in depositional areas. In 2002, EPA collected four-foot sediment core samples from 4 locations in the Wells G and H wetland as part of the Aberjona River Study (Operable Unit 3 [OU-3]). One of these four foot core samples encountered saturated sand before reaching 4 feet (approximately 2 to 3 feet) whereas the remaining cores encountered peat and/or organic deposits as deep as 4 feet, thus documenting the variability in peat thickness/depth.

6.9 Hair Sampling

A study by Rogers et al (1997) analyzed archived human hair samples extending back to 1938 and showed that arsenic content in hair was not correlated with exposure to Wells G and H water. It was anticipated that if high concentrations of arsenic were present in drinking water then the hair analysis would also show elevated levels. The lack of strong correlation indicates that arsenic may not have been present in high concentrations in the water supply.

However, there were likely several opportunities for direct or indirect arsenic removal from the water supply system prior to delivery to individual taps. Water conditioning treatments or reactions during residence in the water distribution system could limit concentrations of arsenic consumed by individuals.

6.10 Other Issues

Note that various organic complexes of arsenic are possible in solution. Organic forms of arsenic are usually minor in surface waters. Nonetheless, proportions of organic forms of arsenic can increase as a result of methylation reactions catalyzed by microbial activity (bacteria, yeasts, algae). The dominant organic forms generally found are DMAA and MMAA, where arsenic is present in both cases in the As(V) state. Proportions of these two species have been noted to increase in summer as a result of increased microbial activity. The organic species may also be more prevalent close to the sediment-water interface (Smedley and Kinniburgh, 2004). In addition, these organic forms of arsenic were found to be inconsequential in groundwater of the Industri-Plex Superfund Site north of Wells G and H in the Aberjona watershed since they were found not to control mobility (Ford, 2004d) and thus may not play a significant role at Wells G and H. However, there are no direct indicators of the potential for organic arsenic species; therefore, there is no basis to exclude the potential for the presence of organic arsenic species within the Wells G and H aquifer (Ford, 2004d).

In groundwaters, the ratio of As(III) to As(V) can vary enormously as a result of large variations in aquifer redox conditions, redox gradients and history. In strongly reducing aquifers (Fe(III)- and sulfate-reducing), As(III) typically dominates. Reducing arsenic-rich groundwaters from Bangladesh have As(III)/Total Arsenic (As_T) ratios varying between 0.1-0.9, but are typically around 0.5-0.6. Ratios in reducing groundwaters from Inner Mongolia are typically 0.7-0.9. Concentrations of organic arsenic species are generally low or negligible in groundwaters (see Smedley and Kinniburgh, 2004).

Redox reactions are important for controlling the behavior of many major and minor species in natural waters, including arsenic. However, in practice, redox equilibrium is often achieved only

slowly. For example, Smedley and Kinniburgh (2004) describe research that estimated that the complete reductive dissolution of Fe(III) oxides in an anoxic Swiss lake sediment would take more than 1000 years. Equilibrium thermodynamic calculations predict that As(V) should dominate over As(III) in all but strongly reducing conditions, i.e. at least sulfate-reducing conditions. However, such theoretical behavior is not necessarily followed quantitatively in natural waters where different redox couples can point to different implied redox potentials (Eh values) reflecting thermodynamic disequilibrium (Smedley and Kinniburgh, 2004). In Oslofjord, Norway, As(III) was found under oxidizing conditions. Also, in oxygenated seawater, the As(V)/As(III) ratios should be of the order of 10¹⁵-10²⁶, whereas measured ratios of 0.1-250 have been found largely supported by biological transformations. Oxidation of As(III) by dissolved oxygen, so-called oxygenation, is a particularly slow reaction. Half-lives for the oxygenation of As(III) in seawater range from several months to a year (Smedley and Kinniburgh, 2004).

Other studies have demonstrated the stability of As(V)/As(III) ratios over periods of days or weeks during water sampling when no particular care was taken to prevent oxidation again suggesting relatively slow oxidation rates. Smedley and Kinniburgh (2004) cite research that found stable ratios in seawater for up to 10 days (4°C). Cherry et al. (1979) found from experimental studies that the As(V)/As(III) ratios were stable in anoxic solutions for up to 3 weeks, but that gradual changes occurred over longer timescales. Cherry et al (1979) also suggested that the measured As(V)/As(III) ratios in natural waters might be used as an indicator of the ambient redox (Eh) conditions, particularly in groundwater where equilibration times are long. Other studies cited by Smedley and Kinniburgh (2004) have also concluded that the As(V)/As(III) ratio may be used as a reliable redox indicator for groundwater systems. However, others have found that the Eh calculated from the As(V)-As(III) couple neither agreed with that from the Fe(II)-Fe(III) and other redox couples nor with the measured Eh. Therefore, the reliability of the arsenic redox couple as a redox indicator remains to be seen. The most that can be said at present is that the existence of As(III) implies reducing conditions somewhere in the system (Smedley and Kinniburgh, 2004).

7.0 CONCLUSIONS & RECOMMENDATIONS

The Technical Memorandum evaluated the potential impact of inorganic contaminants, particularly arsenic, in Aberjona River surface water and Wells G and H wetland sediments on potable water supply development in the Wells G and H Central Area aquifer. This analysis was based on relevant technical papers, site-specific reports, and existing data available for the Wells G and H site.

Conceptual Model

A conceptual model for arsenic migration at the Wells G and H site is illustrated in Figure 20. The conceptual model reflects elevated levels of arsenic present in the shallow wetland peat deposits (source layer) and depicts the principal direction of groundwater flow in an East-West cross-section (toward the center of the river valley). Figure 21 provides representative groundwater elevation contours in plan view. The peat deposit source (see Zones A and the Sp and P2 layers of Zone B in Figure 20) was likely caused by historic releases of arsenic, mainly from the Industri-Plex Superfund Site to the north. Wetland sediment in the specific area of Wells G and H may have as much as 700 kilograms (kg) of arsenic present (Aurilio, 1992), with concentrations ranging from about 500 to 1000 milligrams per kilogram (mg/kg) (Knox, 1991). These concentrations are consistent with the recent Aberjona River Study (M&E, 2004), which documented arsenic concentrations in sediment ranging from 2.4 mg/kg to 4,550 mg/kg in Aberjona River sediments. The Aberjona River Study (M&E, 2004) documented nearly 100-percent detection frequency of arsenic in 325 sediment samples collected along six miles of the river from Route 128 in Woburn to the Mystic Lakes in Arlington and Medford. At the Wells G and H wetland, the median and average arsenic sediment concentrations measured were approximately 105.5 mg/kg and 415 mg/kg, respectively. The major source of arsenic in the Wells G and H wetland appears to be the former industrial activities and contaminant releases at the Industri-Plex Superfund Site.

In the vicinity of the Wells G and H wetland near the Aberjona River, arsenic is highly persistent in the peat with the greatest proportion of arsenic in the upper 80 centimeters of peat. Arsenic is also found in interbedded sand layers within the peat and in deeper peat sediments to a depth of at least 120 to 160 centimeters (Zeeb, 1996). Beneath the peat is a thick (up to 15 feet) diatomaceous silt layer thickening to the east near well H (Zone B, layer D). In other locations within the wetland the peat is directly underlain by glacial drift. The thickness of the peat is variable. For example, in 2002 EPA collected four-foot sediment core samples from 4 locations in the Wells G and H wetland. One of the cores encountered sand and gravel at approximately 2 to 3 feet. The remaining cores encountered peat as deep as 4 feet.

The peat layer has the properties of high organic content, variable oxidation state (aerobic to anaerobic), and relatively high levels of iron (1 to 25-percent by weight; Keon et al, 2001) and sulfur (0.2 to 11-percent by weight) (Ford, 2004b) as well as very low hydraulic conductivity 2×10^{-6} and 6×10^{-4} cm/sec (Bialon, 1995) (see Zone A and the Sp and P2 layers of Zone B in Figure 20). The diatomaceous silt layer should also have a low hydraulic conductivity in the range of 3×10^{-5} cm/sec based on comparable values for other silt layers (McWhorter and Sunada, 1977). Results for iron from the above-mentioned EPA 2002 sediment cores were relatively

consistent with Keon et al (2001), with maximum concentrations of 12-percent by weight (average 3.1-percent by weight). In addition, peat from the EPA cores demonstrated high TOC concentrations (concentration range of 17 to 32-percent with an average concentration of 23-percent by weight).

Below this arsenic rich peat layer is a thick (82 feet) glacial deposit aquifer consisting of stratified drift sediments and till the aquifer is underlain by bedrock (see Zones C and D in Figure 20). The glacial sediments are much lower in organic carbon, iron, and sulfur compared to the peat sediments in Zones A and B (Figure 20). The glacial aquifer is highly permeable with a hydraulic conductivity ranging from 5×10^{-2} to 8×10^{-2} cm/sec (USGS, 1987). The aquifer is well flushed and the groundwater is expected to be relatively young throughout due to the highly permeable nature of the glacial drift sediments, the high recharge rate, and the discharge of groundwater from the bedrock valley basin. The Phase IA report (GeoTrans, 1994) drew few conclusions about tritiated groundwater data. The presence of tritiated groundwater at 480 feet bgs after only 41 years since tritium entered the atmosphere during atomic weapons testing suggests that the groundwater must be relatively young. The DO levels and ORP readings indicate mildly reducing to well oxygenated conditions. The deep groundwater is typified by arsenic groundwater concentrations ranging from 0.22 to 60 ug/L, with a median value of less than 3 ug/L.

Interaction of Surface Water, Sediment and Groundwater

There is evidence to suggest that past pumping at municipal wells G and H caused some induced infiltration drainage of surface water from the Aberjona River to the well field. Specific conditions reported for Wells G & H may have enhanced the potential for induced infiltration from the Aberjona, i.e., a) the shallow gravel packing of these wells, and b) the excavation of peat surrounding these wells and subsequent emplacement of sand and gravel in a 30 foot radius around the wells. Thus, the wells may have induced infiltration of river water, or experienced preferential flow from the river. This would account for the rapid decline in water levels in the wetland observed by the USGS during the 1985/1986 aquifer pump test. It is unlikely that the wetlands could have experienced so sudden a decline given the relatively low permeability of the peat without having a preferential pathway conduit to the well field. In some places the peat and silt sediments may be thinner; in these locations, preferential flow may also be contributing to a more rapid decline in the wetland layers. Furthermore, the USGS (1987) determined that approximately 51-percent of the water received at the wells was derived from Aberjona River streamflow. The other 49-percent would be derived from the large cone of influence (3,000 feet by 1,700 feet oblong shape). The zone of influence of Wells G and H after 30-days of pumping was illustrated previously in Figure 10.

Hypothesis for Arsenic Transport from the Aberjona River

Arsenic concentrations in the water contributed to Wells G and H directly from the river would be affected by filtration of the aquifer material (both mechanical filtration of particulates and sorption reactions with aquifer solids) and geochemical interactions. In the vicinity of Wells G and H, the dissolved arsenic concentration in river water is about 6 ug/L on average as noted during baseflow at Aberjona River sampling stations TT#4 (located at HBHA discharge control

structure at Mishawum Road) and TT#5 (located downstream of Wells G and H at the Salem Street bridge) [TTNUS, 2002]. However, arsenic concentrations may also change through the degree of filtering and geochemical interactions that occur in the aquifer. The water that enters the wells would contain a mix of water infiltrating through the peat layer to the underlying aquifer as well as glacial drift aquifer water that has not passed through the peat deposits. Note that the induced infiltration of the peat would likely diminish over time as the peat layer becomes partially dewatered under the influence of pumping, which would result in significant dilution of any chemicals released from the peat layer. The high flow rate of the aquifer would also promote the migration of more recent meteoric water with higher oxygen levels to the well, which would tend to attenuate the mobility of arsenic because conditions would favor the less mobile and more readily adsorbed As(V) form. Induced infiltration under pumping would also introduce dilute (relative to arsenic) and well oxygenated/highly oxidized river water to the aquifer, leading to a shift in redox equilibria in the aquifer along flow paths toward less reducing/more oxidized conditions. Thus, under extended pumping conditions, arsenic concentrations in the pumped water would be expected to decline due to a combination of dilution and increased adsorption of arsenic.

It is possible that the peat layer may serve as a source of arsenic through direct drainage to the wells. Although additional contributions of arsenic could be derived from any water within the capture zone of the wells that pass through the arsenic-rich sediments, where arsenic has been deposited from upstream industrial activity since the 1800s. The peat layer has high levels of dissolved arsenic in pore water (215 to 674 ug/L) as noted by Ford (2004b). The high levels of arsenic in the peat layer are attributed to anaerobic conditions, which cause dissolution of iron oxides and release of arsenic. (Under aerobic conditions arsenic, especially As(V), tends to strongly sorb to iron oxyhydroxide solids.) If a hydraulic gradient is placed on the pore water, then the arsenic may begin to migrate especially as arsenite (As[III]).

Ford (2004b) conducted oxic leaching tests to assess arsenic release under oxic conditions using a leaching solution with a chemical composition consistent with surface water. Sediment pore water measurements and oxic leach test results indicate that sediments may be a long-term source of arsenic to groundwater at Wells G and H. Arsenic concentrations released from sediments collected from the Wells G and H wetland ranged from 23 µg/L to 341 µg/L. Ford (2005a) notes that the batch tests may not represent a continuous flow system and do not capture conditions potentially induced by microbial activity. Ford (2004b) also notes, however, that the greatest risk of arsenic release from sediments occurs under reducing conditions. Under a pumping scenario where aqueous geochemical conditions become increasingly dependent on the influence of surface water infiltration, the aquifer becomes more oxidized; as a result, steady concentrations like that found in the batch oxic leach tests are unlikely to be sustainable. This outcome is dependent, however, on the degree of mixing of various water sources within the aquifer and would be influenced by microbial activity (Ford 2005a).

Studies regarding arsenite mobility indicate that arsenite is more mobile than arsenate and that under anaerobic conditions both are more mobile than under aerobic conditions (Isenbeck-Schroter et al, 2003). The peat layer contamination could therefore be transported through more transmissive silt layers or possibly through interbedded sand layers provided a window of favorable redox conditions exist.

However, it is unlikely that movement from the shallow peat would lend to any significant arsenic transport. The presence of elevated sulfur and iron conditions in the shallow peat layer sediment may lead to the formation of iron sulfide solids (precipitate) under anaerobic conditions. If the pore water from the peat traveled to the diatomaceous silt or stratified glacial drift aquifer, then under the more oxidizing conditions below the peat, the presence of iron oxide solids would hinder the mobilization of arsenic. Under anaerobic conditions the As(III) migration would dominate and could potentially leach at concentrations greater than the MCL, but the relative quantity of flow would be limited due to the low conductivity of the peat.

The glacial aquifer likely has conditions favoring formation of As(V) which is less mobile and more highly sorbed than other forms. Within the glacial aquifer, organic material is limited and was likely deposited by fluvial processes during the glacial retreat (see Zone D in Figure 20). The limited organic material has likely been oxidized, leaving less reactive cellulose materials, which are not an ideal driver for microbial metabolism. (Iron oxyhydroxide reduction and dissolution is facilitated by microbial activity.) Also, the ORP readings are never more than mildly reducing (-94 mV to -168 mV) in the shallow aquifer, with areas of groundwater exhibiting oxidizing conditions ranging as high as 270 to 556 mV in upland areas approaching the sides of the buried bedrock valley and in the middle and deep portions of the aquifer. Under the Eh-pH conditions generally present in the Central Area Aquifer (see Figure 5), the majority of arsenic is expected to be present as the less mobile, more readily adsorbed As(V) species. In addition, without significant organic material present, only limited microbial reduction of iron oxyhydroxides can take place, which should leave significant iron oxyhydroxide-related adsorptive capacity throughout the aquifer to provide an arsenic sink. Further, Eh-pH conditions favor a significant fraction of iron oxyhydroxy solids (see Figure 3) and the concentration of iron in groundwater is several orders of magnitude greater than that of arsenic, suggesting the potential for a significant presence of iron solids. Hydrous iron oxides have high adsorptive capacity for arsenic and high specific surface area, and therefore are exceptional sinks for arsenic. Consequently, iron oxyhydroxides are a major control on arsenic solubility and mobility.

Summary of Findings

Despite the absence of arsenic data for the period when wells G and H were continuously pumped for potable water supply, there is evidence that the arsenic flux from the Aberjona River, peat sediment infiltration, and the larger glacial drift aquifer were not significant. Data are available for the wells shortly after pumping stopped and during the reactivation of the wells by the USGS in 1985/1986 for completion of an aquifer pump test. Arsenic was detected in the analysis of September 1979 groundwater samples from wells G and H, but at concentrations nearly an order of magnitude lower than the current EPA MCL of 10 ug/L. Also, no sampling of municipal Wells G and H conducted since has detected any arsenic, including sampling conducted during, and at the conclusion of, the USGS 30-day pump test.

Arsenic is relatively persistent once it migrates to an area, since it does not degrade or volatilize. If a high concentration of arsenic (e.g., 50 ug/L, Welch et al, 2000) were detected in the wells during pumping, then it is likely that the arsenic would still be present in the wells. Other

contaminants such as chlorinated solvents, which do degrade, were still present in the wells shortly after the pumping was completed and were again detected during the USGS 1985/1986 aquifer pump test. Arsenic would have been expected to follow a similar pattern had it been present at elevated levels as a result of pumping. Further evidence supporting this supposition that arsenic concentrations were not significant in Wells G and H comes from the J.J. Riley Tannery production wells, which are also located in the Wells G and H area, and were sampled for arsenic during their operational period and arsenic was not detected.

A comprehensive review of arsenic contamination in groundwater was compiled by Smedley and Kinniburgh (2004), who determined concentration ranges for seven water quality parameters that are indicative of high arsenic concentrations. Evaluating the Wells G and H data in comparison to these parameters showed that unfavorable conditions were present in Well G (all 7 parameters unfavorable) and in Well H (5 of 7 parameters unfavorable) for the presence of high arsenic levels (above drinking water standards). The parameters evaluated included iron, manganese, alkalinity, chloride, sulfate, nitrate, and pH. This comparative analysis was previously summarized in Table 2. The widespread presence of sulfate in groundwater at relatively high concentrations (>1,000 mg/L) suggests generally oxidizing conditions, which are not conducive to arsenic mobility.

In addition, under nonpumping conditions, the vertical gradients between the peat and underlying aquifer are variable. During storm events or periods of high evapotranspiration there may be a downward gradient through the peat. However, most of the time there is an upward gradient with glacial aquifer groundwater discharging to the Aberjona River. Under these conditions the shallow sediments and peat layer would slowly purge arsenic to the surface water as dissolved and particulate matter. Countering this effect is the downstream transport of arsenic from the Industri-Plex site which slowly adds about 0.1 centimeter per year of contaminated sediment to the river bottom in the Wells G and H area (Zeeb, 1996).

A study by Rogers et al (1997) analyzed archived human hair samples extending back to 1938 and showed that arsenic content in hair was not correlated with exposure to Wells G and H water. It was anticipated that if high concentrations of arsenic were present in drinking water then the hair analysis would also show elevated levels. The lack of strong correlation indicates that arsenic may not have been present in high concentrations in the water supply. However, there were likely several opportunities for direct or indirect arsenic removal from the water supply system prior to delivery to individual taps.

Based on the collective data, the current weight of the evidence indicates that arsenic contamination in the Wells G and H wetland sediments and Aberjona River surface water and related arsenic contamination in the aquifer would not preclude the use of the aquifer for potable water uses.

OTHER METALS

The other metal contaminants considered in the preparation of this Technical Memorandum (chromium, lead, copper, and mercury) are much less mobile than arsenic and tend to show strong partitioning behavior with aquifer solids and/or organic material. Available data from monitoring wells support this conclusion, as summarized below:

Chromium. In the general Eh-pH regime of the Central Area aquifer, chromium is expected to be in the Cr(III) state and will likely exist as Cr_2O_3 solid. In addition, aqueous partitioning coefficients (K_d) for Cr (III) are significantly greater than comparable values for As(V), which is not very mobile. This information suggests that chromium will be relatively immobile in Central Area groundwater.

Lead. Lead is relatively immobile in all matrices due to its strong tendency to be sorbed by iron and manganese oxides and the insolubility of many lead minerals. Lead is effectively removed from water by adsorption to organic matter and clay minerals, precipitation as insoluble salt, and the reaction with hydrous iron and manganese oxide.

In the general Eh-pH regime of the Central Area aquifer, lead is expected to be in the soluble Pb(II) state, but should be readily adsorbed given the extent of Fe and Mn oxides in the Central Area aquifer and the adsorption tendencies expressed by the Langmuir constants.

The anomalous detections of lead at Wells G and H are inconsistent with the known fate and transport behavior of lead and may reflect sampling or analytical error or contamination by pump materials (e.g., bronze or brass parts).

Copper. Copper is strongly adsorbed by organic matter, Fe(III) and Mn oxides, and secondarily by clays. Bodek et al (1988) state that copper exceeds lead in its tendency to be sorbed to solids. Sorption is probably the most important controlling mechanism in determining copper mobility in the environment.

In the general Eh-pH regime of the Central Area aquifer, copper will largely exist in a precipitated phase, and available soluble Cu(II) species should be readily adsorbed to Fe and Mn oxides given the extent of Fe and Mn oxides in the Central Area aquifer. Copper groundwater data collected on behalf of EPA in 2002 using low stress (low flow) purging protocols set forth by EPA Region 1 showed that copper was detected in Central Area groundwater at concentrations approximately 3 orders of magnitude lower than the EPA drinking water action level of 1,300 ug/L.

Mercury. In the general Eh-pH regime of the Central Area aquifer, mercury will largely exist in low solubility phases, and available soluble mercury species should be readily adsorbed to Fe and Mn oxides given the extent of Fe and Mn oxides in the Central Area aquifer. In addition, mercury groundwater data collected on behalf of EPA in 2002 using low stress (low flow) purging protocols show that mercury was rarely detected in Central Area groundwater. Detected concentrations, and laboratory reporting limits for non-detect results, were typically more than an order of magnitude lower than the EPA MCL of 2 ug/L (generally 0.1 ug/L). The detections of

mercury in Wells G and H at the end of the USGS 30-day aquifer pump test are anomalous and are not consistent with mercury concentrations in the aquifer based on recent (2002) low stress (low flow) groundwater sampling conducted on behalf of EPA.

RECOMMENDATIONS

The foregoing analysis is not an endorsement by EPA or EPA's supporting contractors that treatment for arsenic, or other metals, is not required in the event there is a decision to resume using the aquifer as a potable water supply for public distribution. There is some uncertainty in the analysis since the study relied on existing data and studies in a "weight of evidence" approach to evaluate the possible effects of arsenic and heavy metal contaminated sediment on future potable water development in the Central Area aquifer. If the aquifer is utilized in the future for drinking water purposes, then water quality geochemical parameters and metals (e.g., arsenic, chromium, lead) should be monitored and any detected contaminant levels should be dealt with in accordance with current local, state, and federal policies and regulation, which may include treatment during design, startup, and long-term operation to ensure metals do not migrate to the production well(s) above drinking water standards (e.g., arsenic greater than 10 ug/L). If arsenic does migrate to the production well(s), then treatment options would need to be considered for the extracted water prior to public distribution. Note that treatment may need to be considered regardless due to the elevated levels of manganese, as noted by Dufresne-Henry (1978).

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TABLES

Table 5A
Maximum Arsenic Groundwater Data From 0 - 20 Feet
Plotted in Figure 12A

Well	Sample Date	Concentration (ug/L)
AB-1	9/2/1993	13
AB-4SS	12/18/2002	2.5
AB-6SS	12/16/2002	<2.5
B3A	3/14/2002	1.1
BOW13	9/10/1992	13.9
BOW14	9/15/1992	<3
BOW9	9/14/1992	<3
BSSW5	10/1/1992	8.6
BSSW6	10/22/1987	20
BSW2	9/28/1992	<3
BSW7	9/3/1992	3.4
BW-7	3/12/2002	30.6
DP10	5/28/1992	6.4
DP11	5/29/1992	<1.1
DP12	5/29/1992	4.58
DP13	6/2/1992	<1.1
DP14	6/5/1992	3.5
DP18S	6/5/1992	<1.1
DP19	6/5/1992	4.98
DP20	6/8/1992	<1.1
DP21S	6/5/1992	6
DP22	6/4/1992	5.8
DP24S	6/1/1992	1.1
DP26	6/2/1992	14.8
DP29	5/27/1992	6
DP31	6/3/1992	<1.2
DP32	6/8/1992	<1.1
DP35	6/2/1992	<1.1
DP36	5/29/1992	6.5
DP37S	5/27/1992	1.9
DP38	6/1/1992	<1.1
DP39	6/5/1992	<1.1
DP40	6/1/1992	2.4
DP41	6/8/1992	<1.1
DP6S	6/3/1992	<1.1
DP9S	5/26/1992	1.9
EN-1	4/24/2002	0.82
EN-2	4/24/2002	0.91
EN-4	4/24/2002	2.2
GO1S	2/28/1991	2
IUS1	2/15/1991	<1.1
IUS2C	2/14/1991	59.2
IUS3C	2/13/1991	<2.8
MR-1SS	8/31/1993	<4
MR-2SS	8/31/1993	<2
MR-MW-1	11/8/2001	<4.4

Table 5A Continued		
MR-MW-14	11/8/2001	13.2
MR-MW-15	11/6/2001	<4.6
MR-MW-16	11/9/2001	<4.4
MR-MW-17	11/7/2001	<4.6
MR-MW-18S	11/8/2001	4.6
MR-MW-2	11/7/2001	<4.6
MR-MW-20	11/6/2001	<4.6
MR-MW-21	11/9/2001	5.9
MR-MW-4S	10/17/1998	<10
MR-MW-5S	11/5/2001	<4.6
MR-MW-6	11/6/2001	<4.6
MR-MW-7	11/9/2001	7.9
MR-MW-8	11/7/2001	<4.6
MW-001S	4/16/2002	0.95
MW-002S	4/17/2002	2.5
MW-003	4/18/2002	20
MW-004	4/19/2002	44.2
MW-005	4/18/2002	1.9
MW-006	4/22/2002	37.9
MW-007	4/19/2002	15
MW-008	4/18/2002	3.4
MW-009S	4/23/2002	8
MW-010S	4/22/2002	142
MW-011S	4/26/2002	49.2
MW-012	7/10/2002	69.1
MW-013	7/9/2002	8.7
MW-014S	7/10/2002	4
OL-001	3/12/2002	8
OL-002	12/15/1987	574
OL-003	12/15/1987	15
OL-004	12/15/1987	258
OL-005	12/16/1987	24
OL-006	3/15/2002	267
OL-007	3/14/2002	371
OL-008	3/14/2002	87
OL-015	3/14/2002	5
RMW1	12/12/1991	<10
RMW2	12/12/1991	<10
RMW3	12/12/1991	<10
S21	2/25/1991	12
S44	12/2/1980	<10
S63S	2/26/1991	9.6
S64S	2/22/1991	73.4
S7	12/3/1981	49
S71S	2/21/1991	3.1
S77SS	9/22/1992	105
S78S	10/26/1987	<3
S81S	2/21/1991	14.1
S83SS	8/30/1993	<2
S84S	8/20/1991	<10
S87S	8/23/1991	<10

Table 5A Continued		
S88S	3/7/2002	83.6
S90S	8/22/1991	<10
S91S	9/1/1993	6.2
S92S	3/11/2002	10.6
S93S	3/8/2002	0.55
S94S	8/20/1991	<10
S95S	9/24/1992	3.9
TEST-1	7/9/2002	1.9
UC5	2/27/1991	1.8
UG2	8/26/1991	<10
UG4	8/23/1991	<10
W-MW-1S	10/17/1998	11
W-MW-2S	10/17/1998	3
W-MW-3S	10/17/1998	62
W-MW-4S	11/7/2001	<4.4
W-MW-4SS	9/1/1993	<2
W-MW-5S	12/19/2002	<3
W-MW-6S	12/16/2002	<2.5
WB-1SS	12/18/2002	3.5
EPA3	12/16/1987	6.8
S65S	2/15/1991	2.9
Minimum Detected		0.55
Maximum Detected		574
Median		< 4.6
Average		24.04

Table 5B
Maximum Arsenic Groundwater Data From > 20 - 50 Feet
Plotted in Figure 12B

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	<2.5
BCW13	9/9/1992	<3
BOW15	9/30/1992	<3
BOW16	9/16/1992	<3
BSW1	9/18/1992	<3
BSW12	9/8/1992	<3
BSW14	9/15/1992	<3
BSW6	10/2/1992	59.6
BSW9	9/14/1992	3
BUG1	8/31/1993	11.2
BW1	9/18/1992	<3
BW2	9/28/1992	<3
BW3	11/6/2001	<4.6
BW4	11/6/2001	<4.6
BW5	9/30/1992	<3
BW6R	10/5/1992	<3
BW9	9/11/1992	4.4
DP8	6/2/1992	12.6
EPA1	12/16/1987	9.5
EPA2	12/16/1987	18.7
G15D	11/13/1987	10
G3S	5/15/1985	<10
GO1D	5/15/1985	<10
IUS2B	2/14/1991	<5.2
IUS3B	2/12/1991	9.5
MW-001M	4/15/2002	0.92
MW-002M	4/16/2002	0.71
MW-009M	4/23/2002	2.9
MW-010M	4/25/2002	1.9
MW-011M	4/26/2002	1.7
MW-014M	7/10/2002	0.33
OL-2M	7/9/2002	0.89
OL-3M	7/10/2002	0.55
RW3	3/1/1991	3.6
S10	12/3/1991	<10
S11	12/3/1981	<10
S22	2/19/1991	34.2
S38	1/25/1981	<10
S4	12/3/1981	15
S46	11/14/1980	<10
S47	11/14/1980	<10
S5	12/3/1981	<10
S6	12/3/1981	<10
S60	12/3/1981	<10
S63D	2/26/1991	3.9
S64D	5/14/1985	<10
S64M	2/22/1991	2.3

Table 5B Continued		
S65DR	2/25/1991	1.4
S65M	2/20/1991	<1
S66D	8/30/1993	<1.4
S67M	2/19/1991	23.9
S67S	2/19/1991	16.3
S68S	8/21/1991	<10
S69D	2/13/1991	<1
S71D	4/22/1985	<4
S72S	8/21/1991	10
S73D	2/20/1991	4
S73S	2/20/1991	17.9
S74S	4/23/1985	13
S76S(R)	4/17/2002	1.7
S80S	4/8/1985	<6.1
S81M	6/25/1985	10
S82	2/22/1991	21.7
S84M	8/20/1991	<10
S85S	5/14/1985	<10
S86D	8/26/1991	<10
S86S	8/26/1991	<10
S87M	8/23/1991	<10
S88M	3/7/2002	4.3
S89S	10/16/2002	0.49
S90M	8/22/1991	<10
S91M	8/22/1991	50
S92M	3/11/2002	0.4
S93M	3/8/2002	1.4
S94M	8/20/1991	<10
S95M	9/24/1992	4.4
S97D	9/2/1993	<1.4
UC18	2/27/1991	<1
UC6	2/20/1991	10.7
UG2	8/26/1991	<10
UG4	8/22/1991	<10
W-MW-4M	11/7/2001	<4.6
WB-1M	12/19/2002	<2.5
BSW13	9/11/1992	<3
DP7	6/3/1992	2.7
S38A	1/25/1981	<10
Minimum Detected		0.33
Maximum Detected		59.6
Median		< 4.6
Average		5.2

Table 5C
Maximum Arsenic Groundwater Data From > 50 Feet
Plotted in Figure 12C

Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	<2.5
AB-2R	8/31/1993	<6
AB-4M	12/19/2002	<2.5
BCW14	9/29/1992	<3
BUG1	8/31/1993	<1.4
BW12	9/4/1992	<3
BW13	9/9/1992	<3
BW14	9/29/1992	<3
BW2R	9/28/1992	<3
BW5R	10/1/1992	3.8
G36DB	3/1/1991	<1
G3D	5/15/1985	<10
G3DB	4/24/1985	<4
GEO-1	3/18/2002	0.73
GEO-2	3/15/2002	1.7
GO1DB	2/28/1991	3
IUS2A	2/14/1991	42.6
IUS3A	2/12/1991	<4.3
MR-MW-18D	11/8/2001	<4.4
MR-MW-3BR	11/12/2001	<4.4
MR-MW-5D	11/6/2001	<4.6
MW-009D	4/23/2002	0.68
MW-010D	4/25/2002	2
MW-011D	4/26/2002	0.57
MW-01D	4/15/2002	0.75
MW-02D	4/16/2002	0.41
NEP2	11/6/1987	<2.8
NEP3	12/16/1987	<2.8
S39	9/24/1979	1.5
S40	9/24/1979	2
S41	1/25/1981	<10
S67D	2/19/1991	<1
S68D	10/16/2002	0.26
S72D	8/21/1991	10
S72M	8/21/1991	60
S74D	9/2/1993	<1.4
S76D	3/15/2002	3.3
S76M	3/20/2002	0.8
S77D	9/23/1992	<3
S79D	4/18/1985	32
S8	12/3/1981	<10
S80M	4/8/1985	<6.1
S81D	2/21/1991	4.6
S83	4/1/1985	<10
S83M	8/30/1993	<2

Table 5C Continued		
S84D	8/20/1991	<10
S85M	8/23/1991	<10
S87D	10/15/2002	0.63
S88D	3/7/2002	1.4
S89M	8/26/1991	<10
S90D	8/22/1991	<10
S91D	8/21/1991	<10
S92D	3/12/2002	0.7
S93D	8/20/1993	2.8
S94D	8/20/1991	<10
S95D	9/24/1992	<3
UC11	10/17/2002	0.22
UC12	2/14/1991	<10
UC13	2/20/1991	2.1
UC14	2/19/1991	1.5
UC23	2/27/1991	<5
UC7	2/25/1991	1.5
UG4	8/23/1991	<10
W-MW-4D	11/8/2001	<4.4
WB-1R	12/16/2002	<2.5
Minimum Detected		0.22
Maximum Detected		60
Median		< 3
Average		4.37

Table 5D
Maximum Arsenic Low Flow Groundwater Data From 0 - 20 Feet
Plotted in Figure 12D

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	2.5
AB-2SS	12/17/2002	2.5
AB-4SS	12/18/2002	2.5
AB-6SS	12/16/2002	0.37
B3A	3/14/2002	1.1
BW-7	3/12/2002	30.6
EN-1	4/24/2002	0.82
EN-2	4/24/2002	0.91
EN-4	4/24/2002	2.2
MR-MW-1	11/8/2001	4.4
MR-MW-14	11/8/2001	13.2
MR-MW-15	11/6/2001	4.6
MR-MW-16	11/9/2001	4.4
MR-MW-17	11/7/2001	4.6
MR-MW-18S	11/8/2001	4.6
MR-MW-2	11/7/2001	4.6
MR-MW-20	11/6/2001	4.6
MR-MW-21	11/9/2001	5.9
MR-MW-5S	11/5/2001	4.6
MR-MW-6	11/6/2001	4.6
MR-MW-7	11/9/2001	7.9
MR-MW-8	11/7/2001	4.6
MW-001S	4/16/2002	0.95
MW-002S	4/17/2002	2.5
MW-003	4/18/2002	20
MW-004	4/19/2002	44.2
MW-005	4/18/2002	1.9
MW-006	4/22/2002	37.9
MW-007	4/19/2002	15
MW-008	4/18/2002	3.4
MW-009S	4/23/2002	8
MW-010S	4/22/2002	142
MW-011S	4/26/2002	49.2
MW-012	7/10/2002	69.1
MW-013	7/9/2002	8.7
MW-014S	7/10/2002	4
OL-001	3/12/2002	8
OL-002	3/18/2002	3
OL-003	3/18/2002	9
OL-004	3/13/2002	2
OL-005	3/19/2002	9
OL-006	3/15/2002	267
OL-007	3/14/2002	371
OL-008	3/14/2002	87
OL-015	3/14/2002	5

Table 5D Continued		
S63S	10/17/2002	0.78
S83SS	12/19/2002	2.5
S88S	3/7/2002	83.6
S92S	3/11/2002	10.6
S93S	3/8/2002	0.55
TEST-1	7/9/2002	1.9
WB-1SS	12/18/2002	3.5
W-MW-1S	12/18/2002	2.5
W-MW-4S	11/7/2001	4.4
W-MW-5S	12/19/2002	3
W-MW-6S	12/16/2002	2.5
Minimum Detected		0.37
Maximum Detected		371
Median		4.6
Average		25.00

Table 5E
Maximum Arsenic Low Flow Groundwater Data From 20 - 50 Feet
Plotted in Figure 12E

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	2.5
BW3	11/6/2001	4.6
BW4	11/6/2001	4.6
MW-001M	4/15/2002	0.92
MW-002M	4/16/2002	0.71
MW-009M	4/23/2002	2.9
MW-010M	4/25/2002	1.9
MW-011M	4/26/2002	1.7
MW-014M	7/10/2002	0.33
OL-2M	7/9/2002	1
OL-3M	7/10/2002	1
S72S	3/13/2002	4.9
S73D	3/6/2002	3.2
S73S	3/6/2002	3
S76S(R)	4/17/2002	1.7
S82	10/15/2002	0.13
S85S	10/15/2002	0.1
S86S	10/16/2002	0.9
S88M	3/7/2002	4.3
S89S	10/16/2002	0.49
S92M	3/11/2002	0.4
S93M	3/8/2002	1.4
WB-1M	12/19/2002	2.5
W-MW-4M	11/7/2001	4.6
Minimum Detected		0.1
Maximum Detected		4.9
Median		0.81
Average		2.07

Table 5F		
Maximum Arsenic Low Flow Groundwater Data From > 50 Feet		
Plotted in Figure 12F		
Well	Sample Date	Concentration (ug/L)
AB-4M	12/19/2002	2.5
AB-2M	12/17/2002	2.5
AB-2R	12/17/2002	2.5
GEO-1	3/18/2002	0.73
GEO-2	3/15/2002	1.7
MR-MW-18D	11/8/2001	4.4
MR-MW-3BR	11/12/2001	4.4
MR-MW-5D	11/6/2001	4.6
MW-001D	4/15/2002	0.75
MW-002D	4/16/2002	0.41
MW-009D	4/23/2002	0.68
MW-010D	4/25/2002	2
MW-011D	4/26/2002	0.57
MW-01D	4/15/2002	0.75
MW-02D	4/16/2002	0.41
S68D	10/16/2002	0.26
S72D	3/13/2002	2.6
S72M	3/13/2002	4
S76D	3/15/2002	3.3
S76M	3/20/2002	0.8
S83	12/19/2002	2.5
S87D	10/15/2002	0.63
S88D	3/7/2002	1.4
S91D	3/11/2002	1.2
S92D	3/12/2002	0.7
S93D	3/8/2002	0.87
UC-11	10/17/2002	0.22
WB-1R	12/16/2002	2.5
Minimum Detected		0.22
Maximum Detected		4.6
Median		1.3
Average		1.78

Table 6A
Maximum Chromium Data From 0 - 20 Feet
Plotted in Figure 13A

Well	Sample Date	Concentration (ug/L)
AB-1	12/17/1987	23
AB-2SS	12/17/2002	<2.2
AB-4SS	12/18/2002	5.9
AB-6SS	12/16/2002	1.2
B3A	3/14/2002	<2.5
BOW13	10/20/1987	<3.6
BOW14	9/15/1992	<3.6
BOW9	9/14/1992	<3.6
BSSW5	10/1/1992	17.7
BSSW6	10/22/1987	22
BSW2	9/28/1992	9
BSW7	9/3/1992	4
BW-7	3/12/2002	<2.5
DP10	5/28/1992	12
DP11	5/29/1992	4.4
DP12	5/29/1992	6.7
DP13	6/2/1992	<1
DP14	6/5/1992	<1.4
DP18S	6/5/1992	<1
DP19	6/5/1992	3.6
DP20	6/8/1992	<0.99
DP21S	6/5/1992	19.4
DP22	6/4/1992	15.6
DP24S	6/1/1992	<1
DP26	6/2/1992	52.7
DP29	5/27/1992	1.1
DP31	6/3/1992	<1.1
DP32	6/8/1992	<1
DP35	6/2/1992	<1.1
DP36	5/29/1992	2.2
DP37S	5/27/1992	<1
DP38	6/1/1992	<1
DP39	6/5/1992	<1
DP40	6/1/1992	2.5
DP41	6/8/1992	1
DP6S	6/3/1992	<0.99
DP9S	5/26/1992	3.3
EN-1	4/24/2002	<2.5
EN-2	4/24/2002	<2.5
EN-4	4/24/2002	<2.5
GO1S	2/28/1991	15
IUS1	2/15/1991	<9.4
IUS2C	2/14/1991	21
IUS3C	2/13/1991	15
MR-1SS	8/31/1993	<2

Table 6A Continued

MR-2SS	8/31/1993	<2
MR-MW-1	12/12/2002	0.72
MR-MW-14	11/8/2001	22.6
MR-MW-15	11/6/2001	<1
MR-MW-16	11/9/2001	<4.4
MR-MW-17	11/7/2001	<2.5
MR-MW-18S	12/18/2002	10.3
MR-MW-2	11/7/2001	<2.7
MR-MW-20	11/6/2001	<3.4
MR-MW-21	11/9/2001	24.4
MR-MW-5S	11/5/2001	<1
MR-MW-6	11/6/2001	30.3
MR-MW-7	11/9/2001	<1
MR-MW-8	11/7/2001	<1.1
MW-001S	4/16/2002	<2.5
MW-002S	4/17/2002	11
MW-003	4/18/2002	<2.5
MW-004	4/19/2002	3.7
MW-005	4/18/2002	<2.5
MW-006	4/22/2002	<2.5
MW-007	4/19/2002	<2.5
MW-008	4/18/2002	<2.5
MW-009S	4/23/2002	<2.5
MW-010S	4/22/2002	2.7
MW-011S	4/26/2002	5.8
MW-012	7/10/2002	<2.5
MW-013	7/9/2002	<2.5
MW-014S	7/10/2002	<2.5
OL-001	12/15/1987	281
OL-002	12/15/1987	1220
OL-003	12/15/1987	19
OL-004	12/15/1987	399
OL-005	12/16/1987	12
OL-006	3/15/2002	<3
OL-007	3/14/2002	<3
OL-008	3/14/2002	<3
OL-015	3/14/2002	<3
RMW1	12/12/1991	<10
RMW2	12/12/1991	<10
RMW3	12/12/1991	<10
S21	2/25/1991	93
S4	12/3/1981	16
S44	12/2/1980	53
S63S	2/26/1991	100
S64S	2/22/1991	326
S65S	2/15/1991	71
S7	12/3/1981	22
S71S	2/21/1991	162
S77SS	9/22/1992	7
S78S	10/26/1987	10
S81S	2/21/1991	104

Table 6A Continued		
S83SS	8/30/1993	<4
S84S	8/20/1991	<20
S87S	8/23/1991	<20
S88S	10/7/1992	8
S90S	8/22/1991	<24
S91S	8/21/1991	<20
S92S	3/11/2002	<2.5
S94S	8/20/1991	<25
S95S	9/24/1992	<3.6
TEST-1	7/9/2002	<2.5
UC5	2/27/1991	14
UG2	8/26/1991	<20
UG4	8/23/1991	<20
W-MW-1S	12/18/2002	5.5
W-MW-4S	12/13/2002	3.1
W-MW-4SS	9/1/1993	<4
W-MW-5S	12/19/2002	22
W-MW-6S	12/16/2002	7.4
WB-1SS	9/1/1993	<2
EPA3	12/16/1987	20
S93S	8/27/1991	<21
Minimum Detected		0.72
Maximum Detected		1220
Median		3.85
Average		29.9

Table 6B
Maximum Chromium Groundwater Data From > 20 - 50 Feet
Plotted in Figure 13B

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	<1.5
BCW13	9/9/1992	5
BOW15	9/30/1992	<3.6
BOW16	9/16/1992	<3.6
BSW1	9/18/1992	11
BSW12	9/8/1992	4
BSW14	9/15/1992	<3.6
BSW6	10/20/1987	31
BSW9	9/14/1992	13.4
BUG1	8/31/1993	4.7
BW1	9/18/1992	10
BW2	9/28/1992	9
BW3	9/17/1992	9
BW4	12/19/2002	6.1
BW5	9/30/1992	<3.6
BW6R	10/5/1992	10
BW9	9/11/1992	<3.6
DP8	6/2/1992	14.6
EPA1	12/16/1987	24
EPA2	12/16/1987	12
G15D	11/13/1987	<0.99
G3S	10/29/1987	24
GO1D	2/28/1991	16.8
IUS2B	2/14/1991	37
IUS3B	2/12/1991	50
MW-001M	4/15/2002	<2.5
MW-002M	4/16/2002	<2.5
MW-009M	4/23/2002	<2.5
MW-010M	4/25/2002	<2.5
MW-011M	4/26/2002	<2.5
MW-014M	7/10/2002	<2.5
OL-2M	7/9/2002	<3
OL-3M	7/10/2002	<3
RW3	3/1/1991	5.9
S10	12/3/1991	22
S11	12/3/1981	<10
S22	2/19/1991	148
S38	1/25/1981	<10
S46	11/14/1980	<10
S47	11/14/1980	<10
S5	12/3/1981	<10
S6	12/3/1981	71
S60	12/3/1981	10
S63D	2/26/1991	38
S64D	5/14/1985	34

Table 6B Continued		
S64M	5/14/1985	106
S65DR	2/25/1991	23.3
S65M	2/20/1991	242
S66D	8/30/1993	<2.7
S67M	2/19/1991	102
S67S	2/19/1991	172
S68S	4/23/1985	7.4
S69D	2/13/1991	38
S71D	2/27/1991	16
S72S	11/18/1987	26
S73D	2/20/1991	19
S73S	2/20/1991	93
S74S	4/23/1985	25
S76S(R)	4/17/2002	<2.5
S80S	4/8/1985	<10
S81M	5/14/1985	339
S82	2/22/1991	62
S84M	8/20/1991	<20
S85S	8/23/1991	<20
S86D	8/26/1991	<20
S86S	8/26/1991	<20
S87M	8/23/1991	<20
S88M	10/7/1992	<3.6
S89S	8/26/1991	<21
S90M	8/22/1991	<23
S91M	8/22/1991	40
S92M	3/11/2002	<2.5
S93M	8/27/1991	<20
S94M	8/20/1991	<24
S95M	9/24/1992	12.2
S97D	9/2/1993	8.5
UC18	2/27/1991	<10
UC6	2/20/1991	13
UG2	8/26/1991	<20
UG4	8/22/1991	90
W-MW-4M	9/1/1993	<2
WB-1M	12/19/2002	<2.2
BSW13	9/11/1992	<3.6
DP7	6/3/1992	3.1
Minimum Detected		3.1
Maximum Detected		339
Median		< 10
Average		26.2

Table 6C
Maximum Chromium Data From > 50 Feet
Plotted in Figure 13C

Well	Sample Date	Concentration (ug/L)
AB-2M	8/31/1993	<2
AB-2R	12/17/2002	7
AB-4M	12/19/2002	<1.3
BCW14	9/29/1992	<3.6
BUG1	8/31/1993	2.7
BW12	9/4/1992	<3.6
BW13	9/9/1992	5
BW14	9/29/1992	<3.6
BW2R	9/28/1992	13
BW5R	10/1/1992	<3.6
G36DB	3/1/1991	12
G3D	5/15/1985	<10
G3DB	5/15/1985	10
GEO-1	3/18/2002	<2.5
GEO-2	3/15/2002	<2.5
GO1DB	5/15/1985	<10
IUS2A	2/14/1991	35
IUS3A	2/12/1991	23
MR-MW-18D	11/8/2001	<0.9
MR-MW-3BR	11/12/2001	<1
MR-MW-5D	11/6/2001	133
MW-001D	4/15/2002	<2.5
MW-002D	4/16/2002	<2.5
MW-009D	4/23/2002	<2.5
MW-010D	4/25/2002	<2.5
MW-011D	4/26/2002	<2.5
NEP2	11/6/1987	<9.4
NEP3	12/16/1987	<9.4
S39	8/26/1991	<20
S40	12/24/1985	<25
S41	11/6/1987	<10
S67D	2/19/1991	14
S72D	8/30/1993	8.8
S72M	3/13/2002	<2.5
S74D	2/4/1992	2.7
S76D	3/15/2002	<2.5
S76M	3/20/2002	<2.5
S77D	9/23/1992	10
S79D	4/18/1985	<10
S8	12/3/1981	26
S80M	4/8/1985	<10
S81D	2/21/1991	16
S83	12/19/2002	11
S83M	8/30/1993	<2
S84D	8/20/1991	<20

Table 6C Continued		
S85M	8/23/1991	<20
S87D	8/23/1991	<20
S89D	8/26/1991	<20
S89M	8/26/1991	<20
S90D	8/22/1991	<22
S91D	8/21/1991	<25
S92D	3/12/2002	<2.5
S93D	3/8/2002	<2.5
S94D	8/20/1991	<23
S95D	9/24/1992	<3.6
UC11	2/21/1991	131
UC12	2/14/1991	10600
UC13	2/20/1991	304
UC14	2/19/1991	9040
UC23	2/27/1991	26
UG4	8/23/1991	60
W-MW-4D	11/8/2001	<9.7
WB-1R	12/16/2002	7.8
S68D	4/23/1985	5.7
UC7	2/25/1991	<10
Minimum Detected		2.7
Maximum Detected		10600
Median		< 9.85
Average		304.3

Table 6D
Maximum Chromium Low Flow Groundwater Data From 0 - 20 Feet
Plotted in Figure 13D

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	13
AB-2SS	12/17/2002	2.2
AB-4SS	12/18/2002	5.9
AB-6SS	12/16/2002	1.9
B3A	3/14/2002	2.5
BW-7	3/12/2002	2.5
EN-1	4/24/2002	2.5
EN-2	4/24/2002	2.5
EN-4	4/24/2002	2.5
MR-MW-1	11/8/2001	0.9
MR-MW-14	11/8/2001	22.6
MR-MW-15	11/6/2001	1
MR-MW-16	11/9/2001	4.4
MR-MW-17	11/7/2001	2.5
MR-MW-18S	12/18/2002	10.3
MR-MW-2	11/7/2001	2.7
MR-MW-20	11/6/2001	3.4
MR-MW-21	11/9/2001	24.4
MR-MW-5S	11/5/2001	1
MR-MW-6	11/6/2001	30.3
MR-MW-7	11/9/2001	1
MR-MW-8	11/7/2001	1.1
MW-001S	4/16/2002	2.5
MW-002S	4/17/2002	11
MW-003	4/18/2002	2.5
MW-004	4/19/2002	3.7
MW-005	4/18/2002	2.5
MW-006	4/22/2002	2.5
MW-007	4/19/2002	2.5
MW-008	4/18/2002	2.5
MW-009S	4/23/2002	2.5
MW-010S	4/22/2002	2.7
MW-011S	4/26/2002	5.8
MW-012	7/10/2002	2.5
MW-013	7/9/2002	2.5
MW-014S	7/10/2002	2.5
OL-001	3/12/2002	3
OL-002	3/18/2002	3
OL-003	3/18/2002	3
OL-004	3/13/2002	3
OL-005	3/19/2002	3
OL-006	3/15/2002	3
OL-007	3/14/2002	3
OL-008	3/14/2002	3
OL-015	3/14/2002	3

Table 6D Continued		
S63S	10/17/2002	2.5
S83SS	12/19/2002	1.8
S88S	3/7/2002	2.5
S92S	3/11/2002	2.5
S93S	3/8/2002	2.5
TEST-1	7/9/2002	2.5
WB-1SS	12/18/2002	1.6
W-MW-1S	12/18/2002	5.5
W-MW-4S	12/13/2002	3.1
W-MW-5S	12/19/2002	22
W-MW-6S	12/16/2002	7.4
Minimum Detected		0.9
Maximum Detected		30.3
Median		2.5
Average		4.83

Table 6E
Maximum Chromium Low Flow Groundwater Data From 20 - 50 Feet
Plotted in Figure 13E

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	1.5
BW3	11/6/2001	4.8
BW4	12/19/2002	6.1
MW-001M	4/15/2002	2.5
MW-002M	4/16/2002	2.5
MW-009M	4/23/2002	2.5
MW-010M	4/25/2002	2.5
MW-011M	4/26/2002	2.5
MW-014M	7/10/2002	2.5
OL-2M	7/9/2002	3
OL-3M	7/10/2002	3
S72S	3/13/2002	2.5
S73D	3/6/2002	2.5
S73S	3/6/2002	2.5
S76S(R)	4/17/2002	2.5
S82	10/15/2002	2.5
S85S	10/15/2002	2.5
S86S	10/16/2002	2.5
S88M	3/7/2002	2.5
S89S	10/16/2002	2.5
S92M	3/11/2002	2.5
S93M	3/8/2002	2.5
WB-1M	12/19/2002	2.2
W-MW-4M	11/7/2001	1.5
Minimum Detected		1.5
Maximum Detected		6.1
Median		2.5
Average		2.69

Table 6F		
Maximum Chromium Low Flow Groundwater Data From > 50 Feet		
Plotted in Figure 13F		
Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	1.6
AB-2R	12/17/2002	7
AB-4M	12/19/2002	1.3
GEO-1	3/18/2002	2.5
GEO-2	3/15/2002	2.5
MR-MW-18D	11/8/2001	0.9
MR-MW-3BR	11/12/2001	1
MR-MW-5D	11/6/2001	133
MW-001D	4/15/2002	2.5
MW-002D	4/16/2002	2.5
MW-009D	4/23/2002	2.5
MW-010D	4/25/2002	2.5
MW-011D	4/26/2002	2.5
MW-01D	4/15/2002	2.5
MW-02D	4/16/2002	2.5
S68D	10/16/2002	2.5
S72D	3/13/2002	2.5
S72M	3/13/2002	2.5
S76D	3/15/2002	2.5
S76M	3/20/2002	2.5
S83	12/19/2002	11
S87D	10/15/2002	2.5
S88D	3/7/2002	2.5
S91D	3/11/2002	2.5
S92D	3/12/2002	2.5
S93D	3/8/2002	2.5
UC-11	10/17/2002	12.8
WB-1R	12/16/2002	7.8
Minimum Detected		0.9
Maximum Detected		133
Median		2.5
Average		8.00

Table 7A
Maximum Lead Groundwater Data From 0 - 20 Feet
Plotted in Figure 14A

Well	Sample Date	Concentration (ug/L)
AB-1	12/17/1987	<10
AB-2SS	8/30/1993	<1
AB-4SS	12/18/2002	<4.2
AB-6SS	12/16/2002	<1.4
B3A	3/14/2002	0.3
BOW13	10/20/1987	1.7
BOW14	9/15/1992	<2.8
BOW9	9/14/1992	<2.1
BSSW5	10/1/1992	11
BSSW6	10/22/1987	71
BSW2	9/28/1992	3.7
BSW7	9/3/1992	3.4
BW-7	3/12/2002	0.28
DP10	5/28/1992	8
DP11	5/29/1992	10.1
DP12	5/29/1992	6.2
DP13	6/2/1992	53.8
DP14	6/5/1992	0.99
DP18S	6/5/1992	<0.95
DP19	6/5/1992	2.5
DP20	6/8/1992	2
DP21S	6/5/1992	9.8
DP22	6/4/1992	1.8
DP24S	6/1/1992	<0.97
DP26	6/2/1992	<1.6
DP29	5/27/1992	<0.94
DP31	6/3/1992	<1.2
DP32	6/8/1992	2
DP35	6/2/1992	<1.4
DP36	5/29/1992	<0.99
DP37S	5/27/1992	2.1
DP38	6/1/1992	<0.95
DP39	6/5/1992	<0.95
DP40	6/1/1992	9.5
DP41	6/8/1992	<1.8
DP6S	6/3/1992	69.1
DP9S	5/26/1992	<0.95
EN-1	4/24/2002	<0.25
EN-2	4/24/2002	<0.28
EN-4	4/24/2002	<0.42
GO1S	2/28/1991	17.1
IUS1	2/15/1991	9.2
IUS2C	2/14/1991	25.2
IUS3C	2/13/1991	<35.7
MR-1SS	8/31/1993	<1

Table 7A Continued

MR-2SS	9/1/1993	29
MR-MW-1	11/8/2001	<2
MR-MW-14	11/8/2001	62.7
MR-MW-15	11/6/2001	<2.3
MR-MW-16	11/9/2001	42.5
MR-MW-17	11/7/2001	<2.3
MR-MW-18S	12/18/2002	2.61
MR-MW-2	11/7/2001	<2.3
MR-MW-20	11/6/2001	<2.3
MR-MW-21	11/9/2001	15.5
MR-MW-5S	11/5/2001	<2.3
MR-MW-6	11/6/2001	<2.3
MR-MW-7	11/9/2001	7.8
MR-MW-8	11/7/2001	<2.3
MW-001S	4/16/2002	0.13
MW-002S	4/17/2002	2
MW-003	4/18/2002	0.2
MW-004	4/19/2002	0.28
MW-005	4/18/2002	0.87
MW-006	4/22/2002	<0.19
MW-007	4/19/2002	0.26
MW-008	4/18/2002	0.16
MW-009S	4/23/2002	<0.85
MW-010S	4/22/2002	<0.37
MW-011S	4/26/2002	<0.18
MW-012	7/10/2002	<0.15
MW-013	7/9/2002	<0.18
MW-014S	7/10/2002	<0.11
OL-001	12/15/1987	80
OL-002	12/15/1987	188
OL-003	12/15/1987	<2
OL-004	12/15/1987	192
OL-005	12/16/1987	<2
OL-015	3/14/2002	3
RMW1	12/12/1991	<3
RMW2	12/12/1991	<3
RMW3	12/12/1991	<3
S21	2/25/1991	83
S44	12/2/1980	1460
S63S	10/17/2002	0.97
S64S	2/22/1991	168
S65S	2/15/1991	115
S7	12/3/1981	520
S71S	2/21/1991	50
S77SS	9/22/1992	<1
S78S	10/26/1987	96
S81S	2/21/1991	51
S83SS	12/19/2002	<1.3
S84S	8/20/1991	<5
S87S	8/23/1991	10
S88S	3/7/2002	0.21

Table 7A Continued		
S90S	8/22/1991	9
S91S	9/1/1993	19.7
S92S	3/11/2002	<0.1
S93S	3/8/2002	0.16
S94S	8/20/1991	<5
S95S	9/24/1992	<1
TEST-1	7/9/2002	<0.14
UC5	2/27/1991	51.4
UG2	8/26/1991	<5
UG4	8/23/1991	6
W-MW-1S	12/18/2002	<2
W-MW-4S	11/7/2001	2.1
W-MW-4SS	9/1/1993	<1
W-MW-5S	12/19/2002	<2.1
WB-1SS	12/18/2002	<3.1
Minimum Detection		0.13
Maximum Detection		1460
Median		< 2.3
Average		32.9

Table 7B
Maximum Lead Groundwater Data From > 20 - 50 Feet
Plotted in Figure 14B

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	<1.3
BCW13	9/10/1992	1.8
BOW15	9/30/1992	<1
BOW16	9/16/1992	<1
BSW1	9/18/1992	31.2
BSW12	9/8/1992	<1
BSW14	9/15/1992	<4.8
BSW6	10/20/1987	29
BSW9	9/14/1992	<8
BUG1	8/31/1993	22.6
BW1	9/18/1992	2.8
BW2	9/28/1992	11.4
BW3	11/6/2001	<2.3
BW4	11/6/2001	<2.3
BW5	9/30/1992	<1
BW6R	10/5/1992	<3
BW9	9/11/1992	32
DP8	6/2/1992	18.4
EPA1	12/16/1987	19
EPA2	12/16/1987	16.8
G15D	11/13/1987	12
G3S	10/29/1987	18
GO1D	2/28/1991	7.2
IUS2B	2/14/1991	<5.1
IUS3B	2/12/1991	24.4
MW-001M	4/15/2002	0.69
MW-002M	4/16/2002	5.2
MW-009M	4/23/2002	<0.2
MW-010M	4/25/2002	<0.3
MW-011M	4/26/2002	<0.3
OL-2M	7/9/2002	<0.34
OL-3M	7/10/2002	<0.14
RW3	3/1/1991	13.5
S10	12/3/1991	<40
S11	12/3/1981	<40
S22	2/19/1991	120
S38	1/25/1981	<40
S4	12/3/1981	<40
S46	11/14/1980	<40
S47	11/14/1980	<40
S5	12/3/1981	<4
S6	12/3/1981	43
S60	12/3/1981	<40
S63D	2/26/1991	32.8
S64D	5/14/1985	<5
S64M	2/22/1991	9

Table 7B Continued		
S65DR	2/25/1991	11.9
S65M	2/20/1991	53.5
S66D	8/30/1993	<1.3
S67M	2/19/1991	50
S67S	2/19/1991	40.8
S68S	4/23/1985	50
S69D	2/13/1991	<3.8
S71D	4/22/1985	<50
S72S	11/18/1987	22
S72S	3/13/2002	<0.1
S73D	2/20/1991	13.4
S73S	2/3/1992	5.5
S74S	2/4/1992	2.6
S76S(R)	4/17/2002	<0.1
S80S	4/8/1985	<2.3
S81M	5/14/1985	<5
S82	2/22/1991	35
S85S	8/23/1991	9
S86D	8/26/1991	<5
S86S	10/16/2002	0.16
S87M	8/23/1991	9
S88M	3/7/2002	0.22
S89S	10/16/2002	0.43
S90M	8/22/1991	6
S91M	8/20/1993	54.9
S92M	3/11/2002	0.15
S93M	8/27/1991	<5
S94M	8/20/1991	<5
S95M	9/24/1992	<4.1
S97D	9/2/1993	11.3
UC18	2/27/1991	1.3
UC6	2/20/1991	<0.6
UG2	8/26/1991	10
UG4	8/22/1991	<5
W-MW-4M	11/7/2001	<2.3
WB-1M	12/19/2002	<2.1
BW10	10/20/1987	9.2
DP7	6/3/1992	<2.4
W-MW-4D	11/8/2001	<2
Minimum Detected		0.15
Maximum Detected		120
Median		5.2
Average		12.9

Table 7C
Maximum Lead Groundwater Data From > 50 Feet
Plotted in Figure 14C

Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	<1.3
AB-2R	12/17/2002	<1.5
AB-4M	12/19/2002	<8
BCW14	9/29/1992	<1
BUG1	8/31/1993	2.2
BW12	9/4/1992	3.6
BW13	9/9/1992	2
BW14	9/29/1992	<1
BW2R	9/28/1992	1.1
BW5R	10/1/1992	<1
G36DB	3/1/1991	13
G3D	3/1/1991	7.3
G3DB	5/15/1985	<5
GEO-1	3/18/2002	0.13
GEO-2	3/15/2002	0.53
GO1DB	2/28/1991	2.5
IUS2A	2/14/1991	20.5
IUS3A	2/12/1991	<5.5
MR-MW-18D	11/8/2001	<2
MR-MW-3BR	11/12/2001	<2
MR-MW-5D	11/6/2001	<2.3
MW-001D	4/15/2002	0.17
MW-002D	4/16/2002	<0.1
MW-009D	4/23/2002	<0.12
MW-010D	4/25/2002	<0.4
MW-011D	4/26/2002	<0.21
MW-01D	4/15/2002	0.17
MW-02D	4/16/2002	<0.1
NEP2	11/6/1987	17
NEP3	12/16/1987	<2
S39	12/23/1985	<40
S40	1/25/1981	270
S41	11/6/1987	6.1
S67D	2/19/1991	7.2
S68D	8/21/1991	10
S72D	8/21/1991	6
S72M	8/30/1993	6.3
S74D	2/4/1992	6.9
S76D	3/15/2002	0.67
S76M	3/20/2002	<0.1
S77D	9/23/1992	<1
S79D	4/18/1985	<2.1
S8	12/3/1981	<40
S80M	4/8/1985	3
S81D	2/21/1991	4

Table 7C Continued		
S83	4/23/1985	<50
S83M	8/30/1993	<1
S84D	8/20/1991	<5
S85M	8/23/1991	<5
S87D	8/23/1991	5
S88D	3/7/2002	0.21
S89D	9/2/1993	1.4
S89M	8/26/1991	<5
S90D	8/22/1991	81
S91D	8/21/1991	13
S92D	3/12/2002	0.7
S93D	8/27/1991	15
S94D	8/20/1991	5
S95D	9/24/1992	<1
UC11	10/17/2002	0.85
UC12	2/13/1991	<4.3
UC13	2/20/1991	124
UC14	2/19/1991	48.9
UC23	2/26/1991	2.9
UC7	2/25/1991	4.3
UG4	8/23/1991	15
W-MW-4D	11/8/2001	<2
WB-1R	12/16/2002	<3.2
Minimum Detected		0.13
Maximum Detected		270
Median		2.95
Average		11.87

Table 7D
Maximum Lead Low Flow Groundwater Data From 0 - 20 Feet
Plotted in Figure 14D

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	1.6
AB-2SS	12/17/2002	1.5
AB-4SS	12/18/2002	4.2
AB-6SS	12/16/2002	1.4
B3A	3/14/2002	0.3
BW-7	3/12/2002	0.28
EN-1	4/24/2002	0.25
EN-2	4/24/2002	0.28
EN-4	4/24/2002	0.42
MR-MW-1	11/8/2001	2
MR-MW-14	11/8/2001	62.7
MR-MW-15	11/6/2001	2.3
MR-MW-16	11/9/2001	42.5
MR-MW-17	11/7/2001	2.3
MR-MW-18S	12/18/2002	2.61
MR-MW-2	11/7/2001	2.3
MR-MW-20	11/6/2001	2.3
MR-MW-21	11/9/2001	15.5
MR-MW-5S	11/5/2001	2.3
MR-MW-6	11/6/2001	2.3
MR-MW-7	11/9/2001	7.8
MR-MW-8	11/7/2001	2.3
MW-001S	4/16/2002	0.13
MW-002S	4/17/2002	2
MW-003	4/18/2002	0.2
MW-004	4/19/2002	0.28
MW-005	4/18/2002	0.87
MW-006	4/22/2002	0.19
MW-007	4/19/2002	0.26
MW-008	4/18/2002	0.16
MW-009S	4/23/2002	0.85
MW-010S	4/22/2002	0.37
MW-011S	4/26/2002	0.18
MW-012	7/10/2002	0.15
MW-013	7/9/2002	0.18
MW-014S	7/10/2002	0.11
OL-001	3/12/2002	0.1
OL-002	3/18/2002	0.1
OL-003	3/18/2002	0.3
OL-004	3/13/2002	0.43
OL-005	3/19/2002	0.1
OL-006	3/15/2002	0.16
OL-007	3/14/2002	0.23
OL-008	3/14/2002	0.3
OL-015	3/14/2002	3

Table 7D Continued		
S63S	10/17/2002	0.97
S83SS	12/19/2002	1.3
S88S	3/7/2002	0.21
S92S	3/11/2002	0.1
S93S	3/8/2002	0.16
TEST-1	7/9/2002	0.14
WB-1SS	12/18/2002	3.1
W-MW-1S	12/18/2002	2
W-MW-4S	11/7/2001	2.1
W-MW-5S	12/19/2002	2.1
W-MW-6S	12/16/2002	1.6
Minimum Detection		0.1
Maximum Detection		62.7
Median		0.64
Average		3.28

Table 7E		
Maximum Lead Low Flow Groundwater Data From 20 - 50 Feet		
Plotted in Figure 14E		
Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	1.3
BW3	11/6/2001	2.3
BW4	11/6/2001	2.3
MW-001M	4/15/2002	0.69
MW-002M	4/16/2002	5.2
MW-009M	4/23/2002	0.2
MW-010M	4/25/2002	0.3
MW-011M	4/26/2002	0.3
MW-014M	7/10/2002	0.1
OL-2M	7/9/2002	0.34
OL-3M	7/10/2002	0.14
S72S	3/13/2002	0.1
S73D	3/6/2002	0.21
S73S	3/6/2002	0.44
S76S(R)	4/17/2002	0.1
S82	10/15/2002	0.12
S85S	10/15/2002	0.1
S86S	10/16/2002	0.16
S88M	3/7/2002	0.22
S89S	10/16/2002	0.43
S92M	3/11/2002	0.15
S93M	3/8/2002	0.1
WB-1M	12/19/2002	2.1
W-MW-4M	11/7/2001	2.3
Minimum Detected		0.1
Maximum Detected		5.2
Median		0.26
Average		0.82

Table 7F
Maximum Lead Low Flow Groundwater Data From > 50 Feet
Plotted in Figure 14F

Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	1.3
AB-2R	12/17/2002	1.5
AB-4M	12/19/2002	8
GEO-1	3/18/2002	0.13
GEO-2	3/15/2002	0.53
MR-MW-18D	11/8/2001	2
MR-MW-3BR	11/12/2001	2
MR-MW-5D	11/6/2001	2.3
MW-001D	4/15/2002	0.17
MW-002D	4/16/2002	0.1
MW-009D	4/23/2002	0.12
MW-010D	4/25/2002	0.4
MW-011D	4/26/2002	0.21
MW-01D	4/15/2002	0.17
MW-02D	4/16/2002	0.1
S68D	10/16/2002	0.25
S72D	3/13/2002	0.33
S72M	3/13/2002	0.1
S76D	3/15/2002	0.67
S76M	3/20/2002	0.1
S83	12/19/2002	20
S87D	10/15/2002	3.2
S88D	3/7/2002	0.21
S91D	3/11/2002	0.51
S92D	3/12/2002	0.7
S93D	3/8/2002	0.43
UC-11	10/17/2002	0.85
WB-1R	12/16/2002	3.2
Minimum Detected		0.1
Maximum Detected		20
Median		0.47
Average		1.77

Table 8A
Maximum Iron Groundwater Data From 0 - 20 Feet
Plotted in Figure 15A

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	19100
AB-2SS	8/30/1993	1380
AB-4SS	12/18/2002	5060
AB-6SS	12/16/2002	407
B3A	3/14/2002	140
BW-7	3/12/2002	11100
EN-1	4/24/2002	1050
EN-2	4/24/2002	340
EN-4	4/24/2002	4900
MR-1SS	8/31/1993	3180
MR-2SS	9/1/1993	134
MR-MW-1	12/12/2002	54.4
MR-MW-18S	12/18/2002	3070
MR-MW-2	11/7/2001	39.9
MR-MW-4S	10/17/1998	35000
MR-MW-5S	11/5/2001	<128
MW-001S	4/16/2002	202
MW-002S	4/17/2002	5640
MW-003	4/18/2002	25000
MW-004	4/19/2002	25200
MW-005	4/18/2002	1570
MW-006	4/22/2002	1970
MW-007	4/19/2002	21300
MW-008	4/18/2002	1720
MW-009S	4/23/2002	5420
MW-010S	4/22/2002	18600
MW-011S	4/26/2002	19900
MW-012	7/10/2002	7860
MW-013	7/9/2002	2340
MW-014S	7/10/2002	845
OL-001	3/12/2002	14500
OL-002	3/18/2002	1740
OL-003	3/18/2002	1800
OL-004	3/13/2002	3160
OL-005	3/19/2002	2470
OL-006	3/15/2002	43700
OL-007	3/14/2002	25100
OL-008	3/14/2002	8020
OL-015	3/14/2002	1410
S63S	10/17/2002	450
S83SS	12/19/2002	<43
S88S	3/7/2002	42200
S91S	9/1/1993	4220
S92S	3/11/2002	1740
S93S	3/8/2002	114

Table 8A Continued		
TEST-1	7/9/2002	406
W-MW-1S	10/17/1998	13500
W-MW-2S	10/17/1998	4200
W-MW-3S	10/17/1998	35000
W-MW-4S	11/7/2001	992
W-MW-4SS	9/1/1993	718
W-MW-5S	12/19/2002	19400
W-MW-6S	12/16/2002	20700
WB-1SS	12/18/2002	20000
Minimum Detected		39.9
Maximum Detected		43700
Median		3115
Average		9039.76

Table 8B		
Maximum Iron Groundwater Data From > 20 - 50 Feet		
Plotted in Figure 15B		
Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	552
BUG1	8/31/1993	48900
BW3	11/6/2001	<120
BW4	12/19/2002	318
MW-001M	4/15/2002	252
MW-002M	4/16/2002	38.1
MW-009M	4/23/2002	7360
MW-010M	4/25/2002	874
MW-011M	4/26/2002	519
MW-014M	7/10/2002	<50
OL-2M	7/9/2002	697
OL-3M	7/10/2002	170
S66D	8/30/1993	28.4
S72S	3/13/2002	15500
S73D	3/6/2002	6400
S73S	3/6/2002	5000
S74S	8/31/1993	22300
S76S(R)	4/17/2002	161
S82	10/15/2002	115
S85S	9/2/1993	1970
S86S	10/16/2002	392
S88M	3/7/2002	7690
S89S	10/16/2002	677
S91M	8/20/1993	18400
S92M	3/11/2002	<25
S93M	3/8/2002	19300
S97D	9/2/1993	4670
W-MW-4M	9/1/1993	<32
WB-1M	12/19/2002	158
Minimum Detected		28.4
Maximum Detected		48900
Median		552
Average		5605.34

Table 8C
Maximum Iron Groundwater Data From > 50 Feet
Plotted in Figure 15C

Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	<43.6
AB-2R	12/17/2002	391
AB-4M	12/19/2002	<51
BUG1	8/31/1993	13100
GEO-2	3/15/2002	1100
MR-MW-18D	11/8/2001	300
MR-MW-5D	11/6/2001	4340
MW-009D	4/23/2002	74.5
MW-010D	4/25/2002	2070
MW-011D	4/26/2002	189
MW-01D	4/15/2002	253
MW-02D	4/16/2002	<25
S68D	10/16/2002	<50
S72D	8/30/1993	8330
S72M	8/30/1993	12500
S74D	9/2/1993	2390
S76D	3/15/2002	1250
S76M	3/20/2002	<25
S83	4/1/1985	2000
S83M	8/30/1993	<26
S85M	9/2/1993	<75
S87D	10/15/2002	78.8
S88D	3/7/2002	49.6
S89D	9/2/1993	<82
S91D	3/11/2002	405
S92D	3/12/2002	205
S93D	8/20/1993	4220
UC11	10/17/2002	2660
W-MW-4D	11/8/2001	2250
WB-1R	12/16/2002	323
GEO-1	3/18/2002	<25
Minimum Detected		49.6
Maximum Detected		13100
Median		300
Average		1892.8

Table 8D
Maximum Iron Low Flow Groundwater Data From 0 - 20 Feet
Plotted in Figure 15D

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	19100
AB-2SS	12/17/2002	1210
AB-4SS	12/18/2002	5060
AB-6SS	12/16/2002	407
B3A	3/14/2002	140
BW-7	3/12/2002	11100
EN-1	4/24/2002	1050
EN-2	4/24/2002	340
EN-4	4/24/2002	4900
MR-MW-1	12/12/2002	54.4
MR-MW-18S	12/18/2002	3070
MR-MW-2	11/7/2001	39.9
MR-MW-5S	11/5/2001	128
MW-001S	4/16/2002	202
MW-002S	4/17/2002	5640
MW-003	4/18/2002	25000
MW-004	4/19/2002	25200
MW-005	4/18/2002	1570
MW-006	4/22/2002	1970
MW-007	4/19/2002	21300
MW-008	4/18/2002	1720
MW-009S	4/23/2002	5420
MW-010S	4/22/2002	18600
MW-011S	4/26/2002	19900
MW-012	7/10/2002	7860
MW-013	7/9/2002	2340
MW-014S	7/10/2002	845
OL-001	3/12/2002	14500
OL-002	3/18/2002	1740
OL-003	3/18/2002	1800
OL-004	3/13/2002	3160
OL-005	3/19/2002	2470
OL-006	3/15/2002	43700
OL-007	3/14/2002	25100
OL-008	3/14/2002	8020
OL-015	3/14/2002	1410
S63S	10/17/2002	450
S83SS	12/19/2002	43
S88S	3/7/2002	42200
S92S	3/11/2002	1740
S93S	3/8/2002	114
TEST-1	7/9/2002	406
WB-1SS	12/18/2002	20000
W-MW-1S	12/18/2002	4940
W-MW-4S	11/7/2001	992

Table 8D Continued		
W-MW-5S	12/19/2002	19400
W-MW-6S	12/16/2002	20700
Minimum Detected		39.9
Maximum Detected		43700
Median		2470
Average		8447.90

Table 8E		
Maximum Iron Low Flow Groundwater Data From 20 - 50 Feet		
Plotted in Figure 15E		
Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	552
BW3	11/6/2001	120
BW4	12/19/2002	318
MW-001M	4/15/2002	252
MW-002M	4/16/2002	38.1
MW-009M	4/23/2002	7360
MW-010M	4/25/2002	874
MW-011M	4/26/2002	519
MW-014M	7/10/2002	50
OL-2M	7/9/2002	697
OL-3M	7/10/2002	170
S72S	3/13/2002	15500
S73D	3/6/2002	6400
S73S	3/6/2002	5000
S76S(R)	4/17/2002	161
S82	10/15/2002	115
S85S	10/15/2002	50
S86S	10/16/2002	392
S88M	3/7/2002	7690
S89S	10/16/2002	677
S92M	3/11/2002	25
S93M	3/8/2002	19300
WB-1M	12/19/2002	158
W-MW-4M	11/7/2001	31.8
Minimum Detected		25
Maximum Detected		19300
Median		355
Average		2768.75

Table 8F		
Maximum Iron Low Flow Groundwater Data From > 50 Feet		
Plotted in Figure 15F		
Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	43.6
AB-2R	12/17/2002	391
AB-4M	12/19/2002	51
GEO-1	3/18/2002	25
GEO-2	3/15/2002	1100
MR-MW-18D	11/8/2001	300
MR-MW-5D	11/6/2001	4340
MW-001D	4/15/2002	253
MW-002D	4/16/2002	25
MW-009D	4/23/2002	74.5
MW-010D	4/25/2002	2070
MW-011D	4/26/2002	189
S68D	10/16/2002	50
S72D	3/13/2002	571
S72M	3/13/2002	9230
S76D	3/15/2002	1250
S76M	3/20/2002	25
S83	12/19/2002	1220
S87D	10/15/2002	78.8
S88D	3/7/2002	49.6
S91D	3/11/2002	405
S92D	3/12/2002	205
S93D	3/8/2002	1620
UC-11	10/17/2002	2660
WB-1R	12/16/2002	323
Minimum Detected		25
Maximum Detected		9230
Median		300
Average		1061.98

Table 9A
Maximum Manganese Groundwater Data From 0 - 20 Feet
Plotted in Figure 16A

Well	Sample Date	Concentration (ug/L)
AB-1	12/17/1987	1020
AB-2SS	8/30/1993	734
AB-6SS	12/16/2002	415
B3A	3/14/2002	125
BW-7	3/12/2002	1000
EN-1	4/24/2002	114
EN-2	4/24/2002	477
EN-4	4/24/2002	587
MR-1SS	8/31/1993	320
MR-2SS	9/1/1993	645
MR-MW-1	11/8/2001	54
MR-MW-18S	11/8/2001	470
MR-MW-2	11/7/2001	3.5
MR-MW-4S	10/17/1998	1520
MR-MW-5S	11/5/2001	201
MW-001S	4/16/2002	124
MW-002S	4/17/2002	595
MW-003	4/18/2002	1220
MW-004	4/19/2002	2770
MW-005	4/18/2002	484
MW-006	4/22/2002	2160
MW-007	4/19/2002	820
MW-008	4/18/2002	1390
MW-009S	4/23/2002	581
MW-010S	4/22/2002	1390
MW-011S	4/26/2002	1820
MW-012	7/10/2002	1100
MW-013	7/9/2002	3890
MW-014S	7/10/2002	9010
OL-001	3/12/2002	432
OL-002	3/18/2002	3960
OL-003	3/18/2002	3310
OL-004	3/13/2002	1820
OL-005	3/19/2002	2070
OL-006	3/15/2002	1330
OL-007	3/14/2002	533
OL-008	3/14/2002	174
OL-015	3/14/2002	1140
S63S	10/17/2002	6.4
S83SS	12/19/2002	116
S88S	3/7/2002	2800
S91S	9/1/1993	8210
S92S	3/11/2002	283
S93S	3/8/2002	223
TEST-1	7/9/2002	5290
W-MW-1S	10/17/1998	1780

Table 9A Continued		
W-MW-2S	10/17/1998	680
W-MW-3S	10/17/1998	1520
W-MW-4S	12/13/2002	75.8
W-MW-4SS	9/1/1993	106
W-MW-5S	12/19/2002	2710
W-MW-6S	12/16/2002	1310
WB-1SS	12/18/2002	737
Minimum Detected		3.5
Maximum Detected		9010
Median		737
Average		1427.5

Table 9B
Maximum Manganese Groundwater Data From > 20 - 50 Feet
Plotted in Figure 16B

Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	106
BUG1	8/31/1993	1290
BW3	11/6/2001	7.2
BW4	11/6/2001	118
MW-001M	4/15/2002	111
MW-002M	4/16/2002	609
MW-009M	4/23/2002	551
MW-010M	4/25/2002	1260
MW-011M	4/26/2002	2280
MW-014M	7/10/2002	87.2
OL-2M	7/9/2002	459
OL-3M	7/10/2002	223
S66D	8/30/1993	3.9
S72S	3/13/2002	604
S73D	3/6/2002	391
S73S	3/6/2002	181
S74S	8/31/1993	39800
S76S(R)	4/17/2002	824
S82	10/15/2002	3.8
S85S	9/2/1993	6800
S86S	10/16/2002	118
S88M	3/7/2002	1340
S89S	10/16/2002	5930
S91M	8/20/1993	2610
S92M	3/11/2002	2.8
S93M	3/8/2002	537
S97D	9/2/1993	15100
W-MW-4M	9/1/1993	85
WB-1M	9/1/1993	279
Minimum Detected		2.8
Maximum Detected		39800
Median		459
Average		2817.6

Table 9C
Maximum Manganese Groundwater Data from > 50 Feet
Plotted in Figure 16C

Well	Sample Date	Concentration (ug/L)
AB-2M	8/31/1993	536
AB-2R	12/17/2002	182
AB-4M	12/19/2002	1800
BUG1	8/31/1993	883
GEO-2	3/15/2002	103
MR-MW-18D	11/8/2001	416
MR-MW-5D	11/6/2001	398
MW-009D	4/23/2002	197
MW-010D	4/25/2002	1140
MW-011D	4/26/2002	225
MW-01D	4/15/2002	434
MW-02D	4/16/2002	707
S68D	10/16/2002	60
S72D	8/30/1993	580
S72M	3/13/2002	958
S74D	9/2/1993	12800
S76D	3/15/2002	144
S76M	3/20/2002	54.5
S83	4/1/1985	740
S83M	8/30/1993	502
S85M	9/2/1993	16800
S87D	10/15/2002	162
S88D	3/7/2002	2150
S89D	9/2/1993	16000
S91D	9/1/1993	12200
S92D	3/12/2002	18.2
S93D	3/8/2002	2650
UC11	10/17/2002	884
W-MW-4D	11/8/2001	127
WB-1R	12/16/2002	204
GEO-1	3/18/2002	92.2
Minimum Detected		18.2
Maximum Detected		16800
Median		502
Average		2391.8

Table 9D
Maximum Manganese Low Flow Groundwater Data From 0 - 20 Feet
Plotted in Figure 16D

Well	Sample Date	Concentration (ug/L)
AB-1	12/16/2002	824
AB-2SS	12/17/2002	283
AB-4SS	12/18/2002	392
AB-6SS	12/16/2002	415
B3A	3/14/2002	125
BW-7	3/12/2002	1000
EN-1	4/24/2002	114
EN-2	4/24/2002	477
EN-4	4/24/2002	587
MR-MW-1	11/8/2001	54
MR-MW-18S	11/8/2001	470
MR-MW-2	11/7/2001	3.5
MR-MW-5S	11/5/2001	201
MW-001S	4/16/2002	124
MW-002S	4/17/2002	595
MW-003	4/18/2002	1220
MW-004	4/19/2002	2770
MW-005	4/18/2002	484
MW-006	4/22/2002	2160
MW-007	4/19/2002	820
MW-008	4/18/2002	1390
MW-009S	4/23/2002	581
MW-010S	4/22/2002	1390
MW-011S	4/26/2002	1820
MW-012	7/10/2002	1100
MW-013	7/9/2002	3890
MW-014S	7/10/2002	9010
OL-001	3/12/2002	432
OL-002	3/18/2002	3960
OL-003	3/18/2002	3310
OL-004	3/13/2002	1820
OL-005	3/19/2002	2070
OL-006	3/15/2002	1330
OL-007	3/14/2002	533
OL-008	3/14/2002	174
OL-015	3/14/2002	1140
S63S	10/17/2002	6.4
S83SS	12/19/2002	116
S88S	3/7/2002	2800
S92S	3/11/2002	283
S93S	3/8/2002	223
TEST-1	7/9/2002	5290
WB-1SS	12/18/2002	737
W-MW-1S	12/18/2002	530
W-MW-4S	12/13/2002	75.8

Table 9D Continued		
W-MW-4S	11/7/2001	30
W-MW-5S	12/19/2002	2710
W-MW-6S	12/16/2002	1310
Minimum Detected		3.5
Maximum Detected		9010
Median		591
Average		1274.58

Table 9E		
Maximum Manganese Low Flow Groundwater Data From 20 - 50 Feet		
Plotted in Figure 16E		
Well	Sample Date	Concentration (ug/L)
AB-6M	12/17/2002	106
BW3	11/6/2001	7.2
BW4	11/6/2001	118
MW-001M	4/15/2002	111
MW-002M	4/16/2002	609
MW-009M	4/23/2002	551
MW-010M	4/25/2002	1260
MW-011M	4/26/2002	2280
MW-014M	7/10/2002	87.2
OL-2M	7/9/2002	459
OL-3M	7/10/2002	223
S72S	3/13/2002	604
S73D	3/6/2002	391
S73S	3/6/2002	181
S76S(R)	4/17/2002	824
S82	10/15/2002	3.8
S85S	10/15/2002	1.5
S86S	10/16/2002	118
S88M	3/7/2002	1340
S89S	10/16/2002	5930
S92M	3/11/2002	2.8
S93M	3/8/2002	537
WB-1M	12/19/2002	233
W-MW-4M	11/7/2001	2.3
Minimum Detected		1.5
Maximum Detected		5930
Median		228
Average		665.83

Table 9F
Maximum Manganese Low Flow Groundwater Data from > 50 Feet
Plotted in Figure 16F

Well	Sample Date	Concentration (ug/L)
AB-2M	12/17/2002	100
AB-2R	12/17/2002	182
AB-4M	12/19/2002	1800
GEO-1	3/18/2002	92.2
GEO-2	3/15/2002	103
MR-MW-18D	11/8/2001	416
MR-MW-5D	11/6/2001	398
MW-001D	4/15/2002	434
MW-002D	4/16/2002	707
MW-009D	4/23/2002	197
MW-010D	4/25/2002	1140
MW-011D	4/26/2002	225
S68D	10/16/2002	60
S72D	3/13/2002	290
S72M	3/13/2002	958
S76D	3/15/2002	144
S76M	3/20/2002	54.5
S83	12/19/2002	13
S87D	10/15/2002	162
S88D	3/7/2002	2150
S91D	3/11/2002	22.1
S92D	3/12/2002	18.2
S93D	3/8/2002	2650
UC-11	10/17/2002	884
WB-1R	12/16/2002	204
Minimum Detected		13
Maximum Detected		2650
Median		204
Average		536.16

Table 10A
Maximum Oxidation Reduction Potential Groundwater Data From 0 - 20 Feet
Plotted in Figure 17A

Well	Sample Date	Result (mV)
AB-1	12/16/2002	68
AB-2SS	12/17/2002	20
AB-4SS	12/18/2002	-26
AB-6SS	12/16/2002	366
B3A	3/14/2002	138.4
BSSW16	12/20/2002	34
BUG1	5/11/1993	-134
EN-1	4/24/2002	121.6
EN-2	4/24/2002	140.8
EN-4	4/24/2002	123.7
MR-MW-1	12/12/2002	136
MR-MW-18S	12/18/2002	18
MR-MW-2	12/13/2002	139
MR-MW-4S	12/13/2002	47
MR-MW-5S	12/12/2002	170
MR-MW-5S	12/19/2002	-98
MW-001S	4/16/2002	424
MW-002S	4/17/2002	557
MW-003	4/18/2002	40
MW-004	4/19/2002	-10
MW-005	4/18/2002	91
MW-006	4/22/2002	86
MW-007	4/19/2002	-26
MW-008	4/18/2002	123
MW-009S	4/23/2002	56
MW-010S	4/22/2002	-46
MW-011S	4/26/2002	15
MW-012	7/10/2002	-93
MW-013	7/9/2002	32
MW-014S	7/9/2002	51
OL-001	3/12/2002	-34
OL-002	3/18/2002	-4
OL-003	3/18/2002	76
OL-004	3/13/2002	90
OL-005	3/19/2002	47
OL-006	3/15/2002	-23
OL-007	3/14/2002	-38
OL-008	3/14/2002	-26
OL-015	3/14/2002	147
S63S	10/17/2002	235.1
S77SS	12/20/2002	-3
S83SS	12/19/2002	262
S84S	8/20/1991	169
S87S	8/23/1991	30
S88S	3/7/2002	-45.3
S90S	8/22/1991	82

Table 10A Continued		
S91S	9/1/1993	127
S92S	3/11/2002	45.3
S93S	3/8/2002	150
S94S	8/20/1991	128
TEST-1	7/9/2002	-62.9
W-MW-1S	12/18/2002	-15
W-MW-6S	12/16/2002	139
WB-1SS	12/18/2002	-73
Minimum Detection		-134
Maximum Detection		557
Median		49
Average		73.5

Table 10B
Maximum Oxidation Reduction Potential Groundwater Data From > 20 - 50 Feet
Plotted in Figure 17B

Well	Sample Date	Result (mV)
AB-6M	12/17/2002	172
BOW16	12/20/2002	256
BUG1	5/11/1993	-113
BW-2	12/19/2002	41
BW-3	12/17/2002	-23
BW-4	12/19/2002	114
BW-7	3/12/2002	33.8
MR-MW-4M	12/20/2002	129
MW-009M	4/23/2002	-158
MW-010M	4/25/2002	101
MW-011M	4/26/2002	-22
MW-014M	7/10/2002	217
MW-01M	4/15/2002	266
MW-02M	4/16/2002	104
OL-2M	7/9/2002	270
OL-3M	7/10/2002	142
S68S	8/21/1991	37
S72S	3/13/2002	-44.1
S73D	3/6/2002	26
S73S	3/6/2002	81.9
S74S	8/31/1993	127
S76SR	4/17/2002	122.2
S77S	12/20/2002	185
S82	10/15/2002	210.9
S84M	8/20/1991	133
S85S	10/15/2002	226
S86D	8/26/1991	10
S86S	10/16/2002	139.1
S87M	8/23/1991	38
S88M	3/7/2002	27.6
S89S	10/16/2002	44.4
S90M	8/22/1991	95
S91M	9/1/1993	103
S92M	3/11/2002	117
S93M	3/8/2002	-20.4
S94M	8/20/1991	250
S97D	9/2/1993	67
WB-1M	12/19/2002	18
Minimum Detection		-158
Maximum Detection		270
Median		102
Average		92.7

Table 10C
Maximum Oxidation Reduction Potential Groundwater Data From > 50 Feet
Plotted in Figure 17C

Well	Sample Date	Result (mV)
AB-2M	12/18/2002	-63
AB-2R	12/17/2002	-161
AB-4M	12/19/2002	246
BUG1	5/10/1993	-128
BW16R	12/20/2002	202
GEO-1	3/18/2002	119.6
GEO-2	3/15/2002	49.2
MR-MW-18D	12/18/2002	-65
MR-MW-4D	12/20/2002	-103
MR-MW-5D	12/12/2002	-196
MW-009D	4/23/2002	109
MW-010D	4/25/2002	-96
MW-011D	4/26/2002	151
MW-01D	4/15/2002	97
MW-02D	4/16/2002	118
S40	8/21/1991	-60
S68D	10/16/2002	207.4
S72D	3/13/2002	-32.3
S72M	3/13/2002	-42.5
S74D	9/2/1993	89
S76D	3/15/2002	138.4
S76M	3/20/2002	230.5
S77D	12/19/2002	151
S77M	12/19/2002	152
S83	12/19/2002	208
S84D	8/20/1991	98
S85M	9/2/1993	87
S87D	10/15/2002	113.5
S88D	3/7/2002	226.3
S89D	9/2/1993	99
S89M	8/26/1991	77
S90D	8/22/1991	79
S91D	3/11/2002	241
S92D	3/12/2002	129
S93D	3/8/2002	56.8
S94D	8/20/1991	255
UC-11	10/17/2002	-192.7
WB-1R	12/16/2002	12
Minimum Detection		-196
Maximum Detection		255
Median		97.5
Average		68.5

Table 11A
Maximum Dissolved Oxygen Groundwater Data From 0 - 20 Feet
Plotted in Figure 18A

Well	Sample Date	Concentration (mg/L)
AB-1	12/16/2002	1.72
AB-2SS	12/17/2002	0.17
AB-4SS	12/18/2002	0.11
AB-6SS	12/16/2002	1.21
B3A	3/14/2002	0.0015
BSSW16	12/20/2002	0.46
BUG1	5/11/1993	0.0005
BUG1	5/11/1993	0.0005
EN-1	4/24/2002	0.0046
EN-2	4/24/2002	0.0077
EN-4	4/24/2002	0.0012
MR-MW-1	12/12/2002	4.08
MR-MW-18S	12/18/2002	0
MR-MW-2	12/13/2002	7.75
MR-MW-4S	12/13/2002	1.03
MR-MW-5S	12/12/2002	1.23
MW-001S	4/16/2002	2.7
MW-002S	4/17/2002	1.7
MW-003	4/18/2002	0
MW-004	4/19/2002	1
MW-005	4/18/2002	0
MW-006	4/22/2002	0
MW-007	4/19/2002	0
MW-008	4/18/2002	2
MW-009S	4/23/2002	2
MW-010S	4/22/2002	0
MW-011S	4/26/2002	5
MW-012	7/10/2002	0
MW-013	7/9/2002	0
MW-014S	7/9/2002	0
OL-001	3/12/2002	0
OL-002	3/18/2002	0
OL-003	3/18/2002	0
OL-004	3/13/2002	0
OL-005	3/19/2002	6
OL-006	3/15/2002	0
OL-007	3/14/2002	0
OL-008	3/14/2002	1
OL-015	3/14/2002	0
S63S	10/17/2002	0.0045
S77SS	12/20/2002	0.22
S83SS	12/19/2002	1.7
S88S	3/7/2002	0.00026
S91S	9/1/1993	0.0026
S92S	3/11/2002	0.00072
S93S	3/8/2002	0.00063

Table 11A Continued		
TEST-1	7/9/2002	0.0025
W-MW-1S	12/18/2002	1.31
W-MW-6S	12/16/2002	0.04
WB-1SS	12/18/2002	1.29
Minimum Detected		0
Maximum Detected		7.75
Median		0.00355
Average		0.87

Table 11B
Maximum Dissolved Oxygen Groundwater Data From > 20 - 50 Feet
Plotted in Figure 18B

Well	Sample Date	Concentration (mg/L)
AB-6M	12/17/2002	0.15
BOW16	12/20/2002	0.26
BUG1	8/31/1993	0.004
BW-2	12/19/2002	0.96
BW-3	12/17/2002	1.92
BW-4	12/19/2002	2.96
BW-7	3/12/2002	0.00051
MR-MW-4M	12/20/2002	1.13
MW-009M	4/23/2002	1
MW-010M	4/25/2002	3
MW-011M	4/26/2002	0
MW-014M	7/10/2002	1
MW-01M	4/15/2002	0
MW-02M	4/16/2002	8.2
OL-2M	7/9/2002	0.00048
OL-3M	7/10/2002	0.00037
S72S	3/13/2002	0.00017
S73D	3/6/2002	0.00058
S73S	3/6/2002	0.0014
S74S	8/31/1993	0.0015
S76SR	4/17/2002	0.00029
S77S	12/20/2002	0.26
S82	10/15/2002	0.0091
S85S	9/2/1993	0.0029
S85S	10/15/2002	0.0062
S86S	10/16/2002	0.0069
S88M	3/7/2002	0.00087
S89S	10/16/2002	0.00022
S91M	9/1/1993	0.0043
S92M	3/11/2002	0.00048
S93M	3/8/2002	0.00077
S97D	9/2/1993	0.0046
WB-1M	12/19/2002	0.81
Minimum Detected		0.00017
Maximum Detected		8.2
Median		0.0043
Average		0.66

Table 11C
Maximum Dissolved Oxygen Groundwater Data From > 50 Feet
Plotted in Figure 18C

Well	Sample Date	Concentration (mg/L)
AB-2M	12/18/2002	0
AB-2R	12/17/2002	1.03
AB-4M	12/19/2002	0.19
BUG1	5/10/1993	0.0002
BUG1	5/10/1993	0.0002
BUG1	5/10/1993	0.0001
BUG1	5/10/1993	0.0002
BUG1	5/11/1993	0.0002
BUG1	8/31/1993	0.0014
BUG1	8/31/1993	0.0013
BW16R	12/20/2002	0.08
GEO-1	3/18/2002	0.00093
GEO-2	3/15/2002	0.00057
MR-MW-18D	12/18/2002	0.36
MR-MW-4D	12/20/2002	0.55
MR-MW-5D	12/12/2002	1.81
MW-009D	4/23/2002	0
MW-010D	4/25/2002	0
MW-011D	4/26/2002	4
MW-01D	4/15/2002	0
MW-02D	4/16/2002	7.9
S68D	10/16/2002	0.0065
S72D	8/30/1993	0.0012
S72D	3/13/2002	0.0011
S72M	8/30/1993	0.0006
S72M	3/13/2002	0.00059
S74D	9/2/1993	0.0038
S76D	3/15/2002	0.0016
S76M	3/20/2002	0.0014
S77D	12/19/2002	0.57
S77M	12/19/2002	0.42
S83	12/19/2002	0.27
S85M	9/2/1993	0.0028
S87D	10/15/2002	0.003
S88D	3/7/2002	0.0012
S89D	9/2/1993	0.0008
S91D	9/1/1993	0.0086
S91D	3/11/2002	0.0025
S92D	3/12/2002	0.00033
S93D	9/1/1993	0.001
S93D	3/8/2002	0.00033
UC-11	10/17/2002	0.00057
Minimum Detected		0.0001
Maximum Detected		7.9
Median		0.0013
Average		0.41

Table 12A
Maximum Sulfate Groundwater Data From 0 - 20 Feet
Plotted in Figure 19A

Well	Sample Date	Concentration (ug/L)
DP10	5/28/1992	36000
DP11	4/7/1993	6200
DP12	5/29/1992	23000
DP13	6/2/1992	10400
DP14	6/5/1992	600
DP18S	6/5/1992	39000
DP19	6/5/1992	25000
DP20	6/5/1992	7700
DP21S	6/5/1992	30000
DP22	6/4/1992	24000
DP24S	6/1/1992	31000
DP26	6/2/1992	34000
DP29	4/7/1993	28000
DP2S	5/7/1992	16100
DP31	4/5/1993	12900
DP32	6/3/1992	38000
DP35	6/8/1992	17200
DP36	4/5/1993	4400
DP37S	5/29/1992	22000
DP38	5/27/1992	41000
DP39	6/1/1992	40000
DP40	6/5/1992	21200
DP41	4/5/1993	5200
DP6S	6/8/1992	24000
DP9S	6/3/1992	21000
GO1S	5/26/1992	16500
IUS1	3/29/1993	36800
IUS2C	2/15/1991	59700
MW-001S	2/14/1991	10000
MW-003	4/16/2002	70000
MW-004	4/18/2002	60000
MW-005	4/19/2002	15000
MW-006	4/18/2002	32000
MW-007	4/22/2002	35000
MW-008	4/19/2002	23000
MW-010S	4/18/2002	78000
MW-012	4/22/2002	10000
MW-013	7/10/2002	29000
MW-014S	7/9/2002	18000
OL-002	7/10/2002	28000
OL-003	3/18/2002	13000
OL-005	3/19/2002	25000
RMW1	12/12/1991	19800
RMW2	12/12/1991	43000
RMW3	12/21/1991	9400

Table 12A Continued		
S21	2/25/1991	23700
S63S	2/26/1991	22300
S64S	2/22/1991	26200
S65S	2/15/1991	25800
S7	12/21/1991	34300
S71S	2/21/1991	25000
S81S	2/21/1991	12100
S84S	8/20/1991	18400
S87S	8/23/1991	28000
S88S	3/7/2002	73000
S90S	8/22/1991	7800
S91S	8/21/1991	49000
S92S	3/11/2002	19000
S93S	8/27/1991	102000
S94S	8/20/1991	19200
TEST-1	7/9/2002	11000
Minimum Detected		600
Maximum Detected		102000
Median		24000
Average		27638

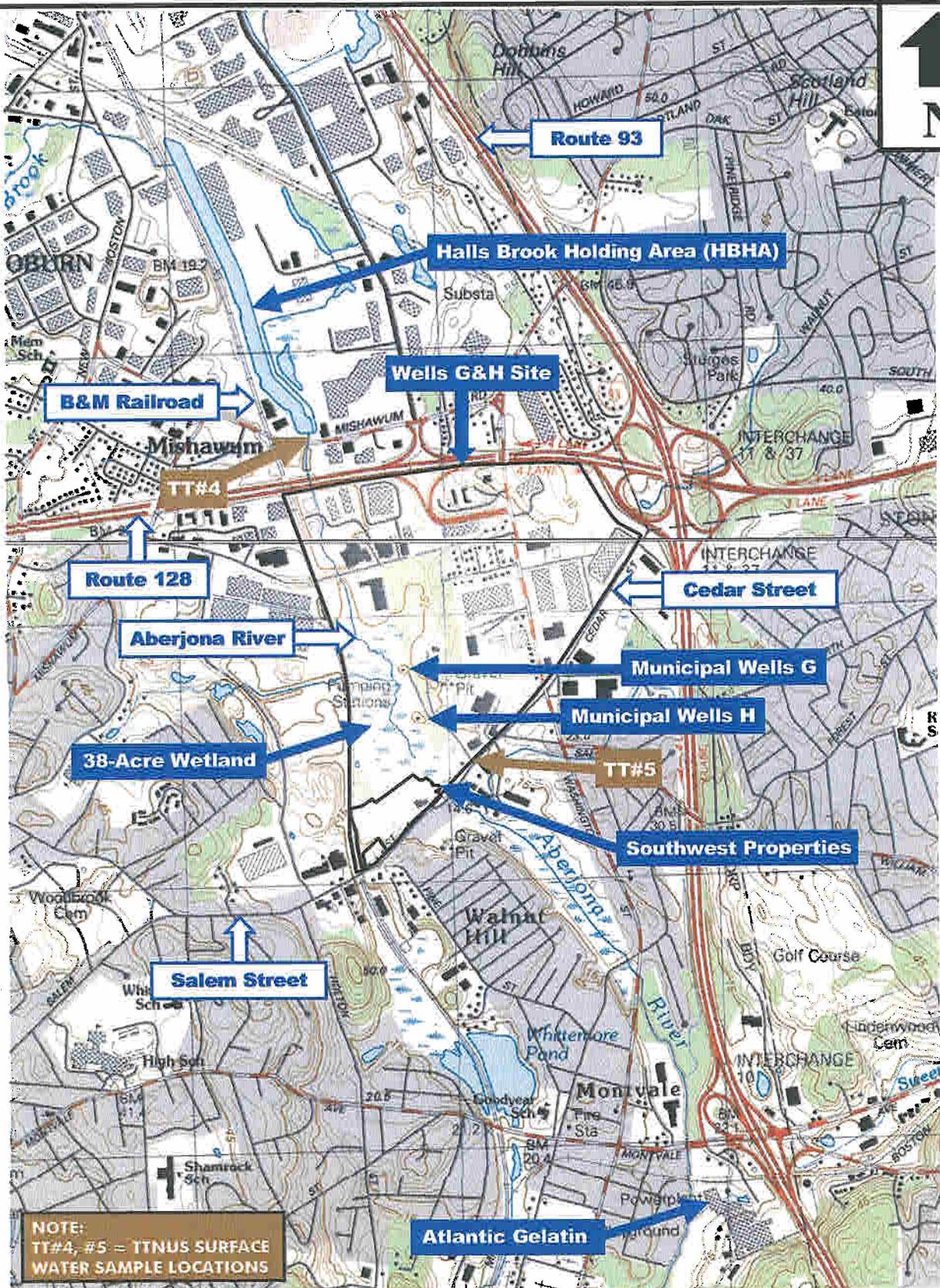
Table 12B
Maximum Sulfate Groundwater Data From 20 - 50 Feet
Plotted in Figure 19B

Well	Sample Date	Concentration (ug/L)
DP2D	04/05/1993	21700
DP2M	04/05/1993	11900
DP7	06/03/1992	35000
DP8	06/02/1992	28000
GO1D	03/29/1993	21400
IUS2B	02/14/1991	84100
MW-010M	04/25/2002	49000
MW-014M	07/10/2002	22000
OL-2M	07/09/2002	33000
OL-3M	07/10/2002	34000
RW3	03/01/1991	9000
S22	04/24/1985	67400
S63D	02/26/1991	26500
S64D	02/25/1991	39500
S64M	02/22/1991	29600
S65DR	02/25/1991	31700
S65M	02/20/1991	23300
S66D	06/05/1992	38000
S67M	02/19/1991	80200
S67S	02/19/1991	39800
S68S	04/23/1985	43700
S69D	02/13/1991	81500
S72S	08/21/1991	30000
S73D	04/23/1985	27300
S73S	02/03/1992	26200
S74S	04/23/1985	49600
S76SR	04/17/2002	52000
S81M	02/21/1991	14200
S82	02/22/1991	17900
S84M	08/20/1991	31000
S85S	08/23/1991	32000
S86D	08/26/1991	25000
S86S	08/26/1991	24000
S87M	08/23/1991	23000
S88M	03/07/2002	44000
S89S	08/26/1991	63000
S90M	08/22/1991	33000
S91M	08/20/1993	35000
S93M	08/27/1991	65000
S94M	08/20/1991	23000
S97D	09/02/1993	46000
Minimum Detected		9000
Maximum Detected		84100
Median		32000
Average		36866

Table 12C
Maximum Sulfate Groundwater Data From > 50 Feet
Plotted in Figure 19C

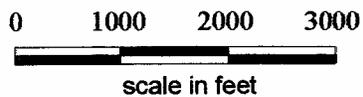
Well	Sample Date	Concentration (ug/L)
G36DB	03/01/1991	39000
G3D	03/01/1991	40000
GEO-1	03/18/2002	40000
GEO-2	03/15/2002	37000
GO1DB	03/29/1993	29200
IUS2A	02/14/1991	56400
MW-010D	04/25/2002	26000
S39	04/13/1976	150000
S40	08/04/1976	143000
S67D	02/19/1991	92100
S68D	04/23/1985	60000
S72D	08/21/1991	19100
S72M	08/30/1993	36000
S74D	04/23/1985	27300
S81D	02/21/1991	11700
S84D	08/20/1991	31000
S85M	08/23/1991	33000
S87D	08/23/1991	32000
S88D	03/07/2002	30000
S89D	09/02/1993	35000
S89M	08/26/1991	33000
S90D	08/22/1991	10700
S91D	09/01/1993	35000
S93D	08/27/1991	49000
S94D	08/20/1991	17300
UC22	03/29/1993	30100
Minimum Detected		10700
Maximum Detected		150000
Median		34000
Average		43958

FIGURES



NOTE:
 TT#4, #5 = TTNUS SURFACE
 WATER SAMPLE LOCATIONS

**BASE MAP IS A PORTION OF THE
 FOLLOWING 7.5' USGS
 TOPOGRAPHIC QUADRANGLE:
 BOSTON NORTH, 1985**



**FIGURE 1
 LOCATION MAP
 WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS**

M&E Metcalf & Eddy

TRC

QUADRANGLE
 LOCATION

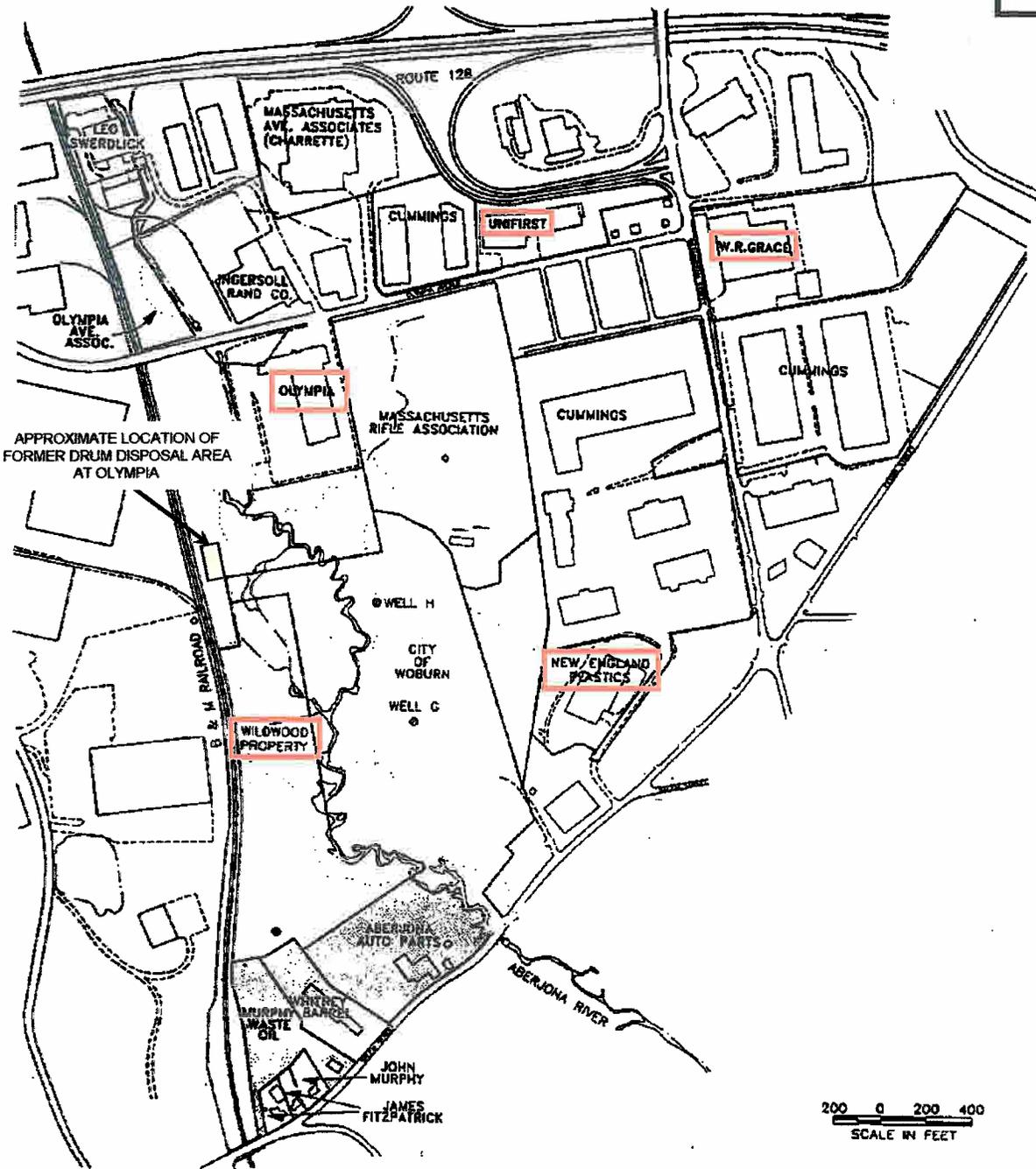


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RAC SUBCONTRACT NO.: 107061



APPROXIMATE LOCATION OF FORMER DRUM DISPOSAL AREA AT OLYMPIA

LEGEND

 **Wells G&H Source Area Properties**

FIGURE 2
SOURCE AREA
PROPERTY LOCATIONS
WELLS G&H
SUPERFUND SITE
WOBURN, MASSACHUSETTS





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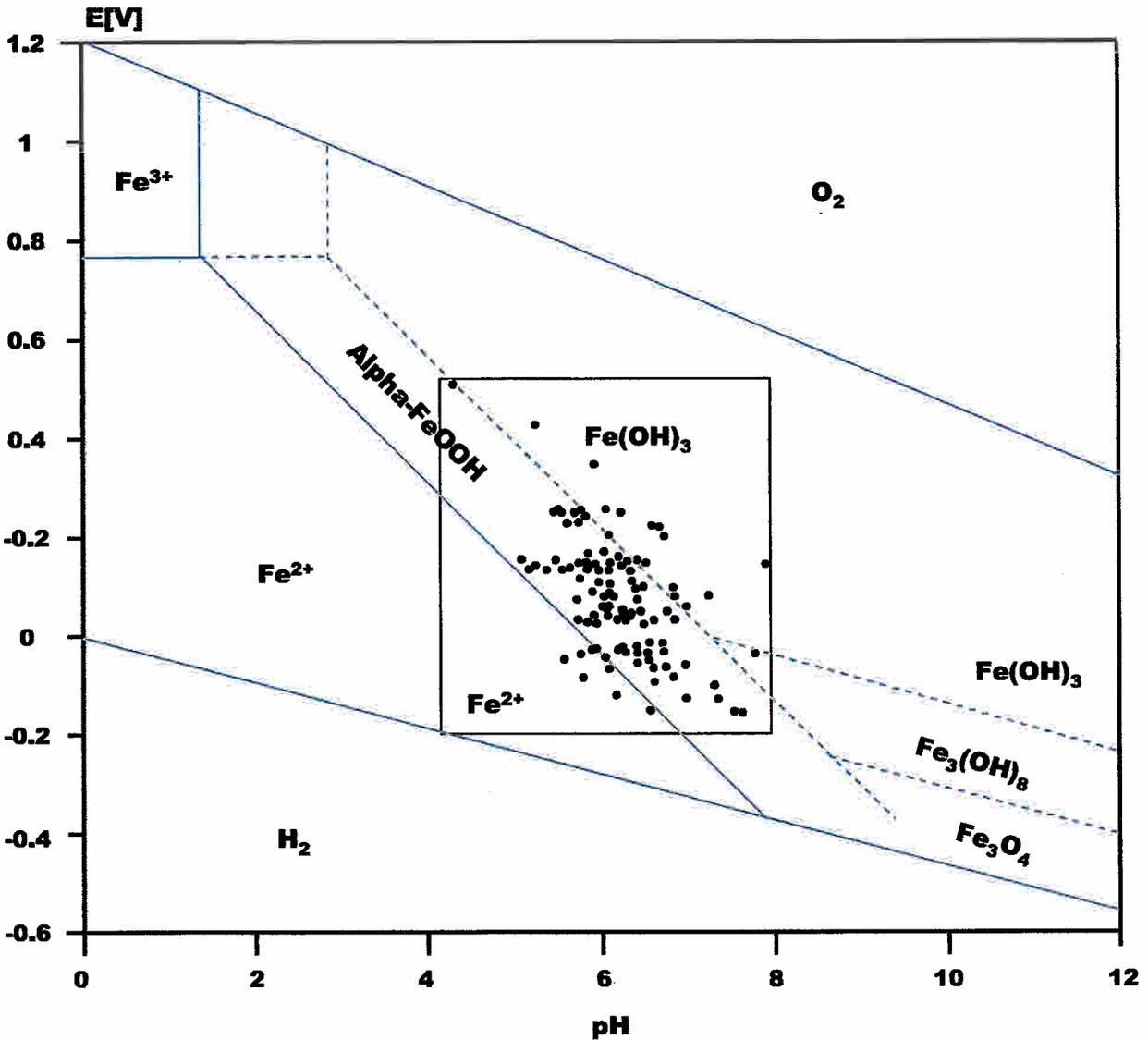
QUADRANGLE LOCATION



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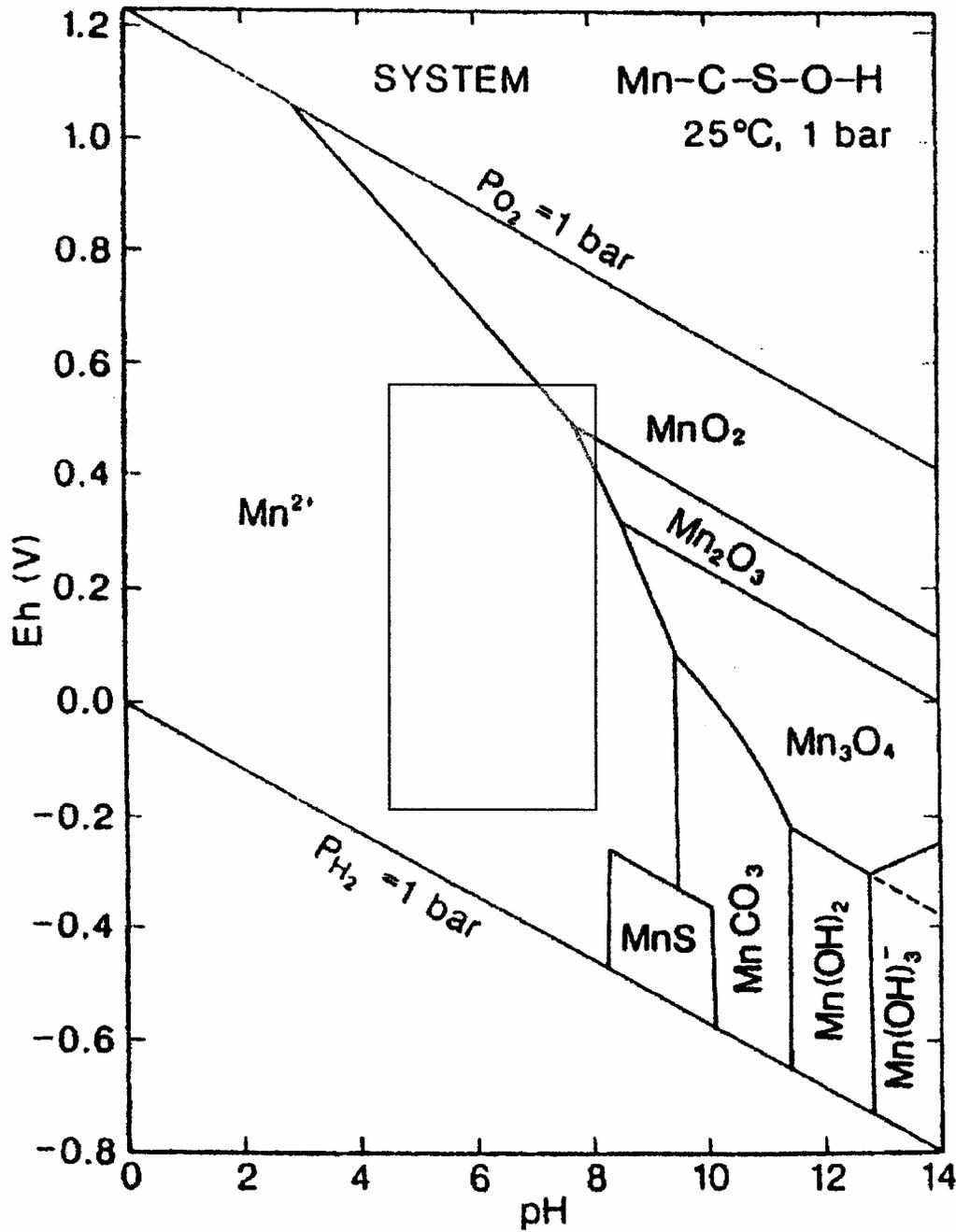


KEY	
	Site Ground Water Eh-pH Regime
	Actual pH and Eh Values from Central Area Aquifer

Reference: Scheffer et al 1989 in: <http://www.soils.wisc.edu/courses/SS325/oxides.htm>

02136WELLS G&H/OU-3/STABILITY

<p>FIGURE 3 Eh-pH STABILITY DIAGRAM WELLS G&H OU1 OLYMPIA PROPERTY SUPERFUND SITE WOBURN, MASSACHUSETTS</p> 	 Boott Mills South Foot of John Street Lowell, MA 01852 978-970-5800
	TRC PROJ. NO.: 02136-0570-01461
	EPA CONTRACT NO.: 68-W6-0042
	RAC SUBCONTRACT NO.: 107961



KEY
 Site Ground Water Eh-pH Regime

Source: ERG, 2003.

FIGURE 4
 SITE Eh-pH REGIME FOR MANGANESE IN
 GROUND WATER SUPERIMPOSED ON Eh-pH DIAGRAM FOR
 THE SYSTEM Mn-O₂-CO₂-S-H₂O, ASSUMING THAT
 $\Sigma Mn = 10^{-3} \text{ mol/kg}$, $\Sigma C = 10^{-3} \text{ mol/kg}$, AND $\Sigma S = 10^{-3} \text{ mol/kg}$
 WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS

M&E Metcalf & Eddy

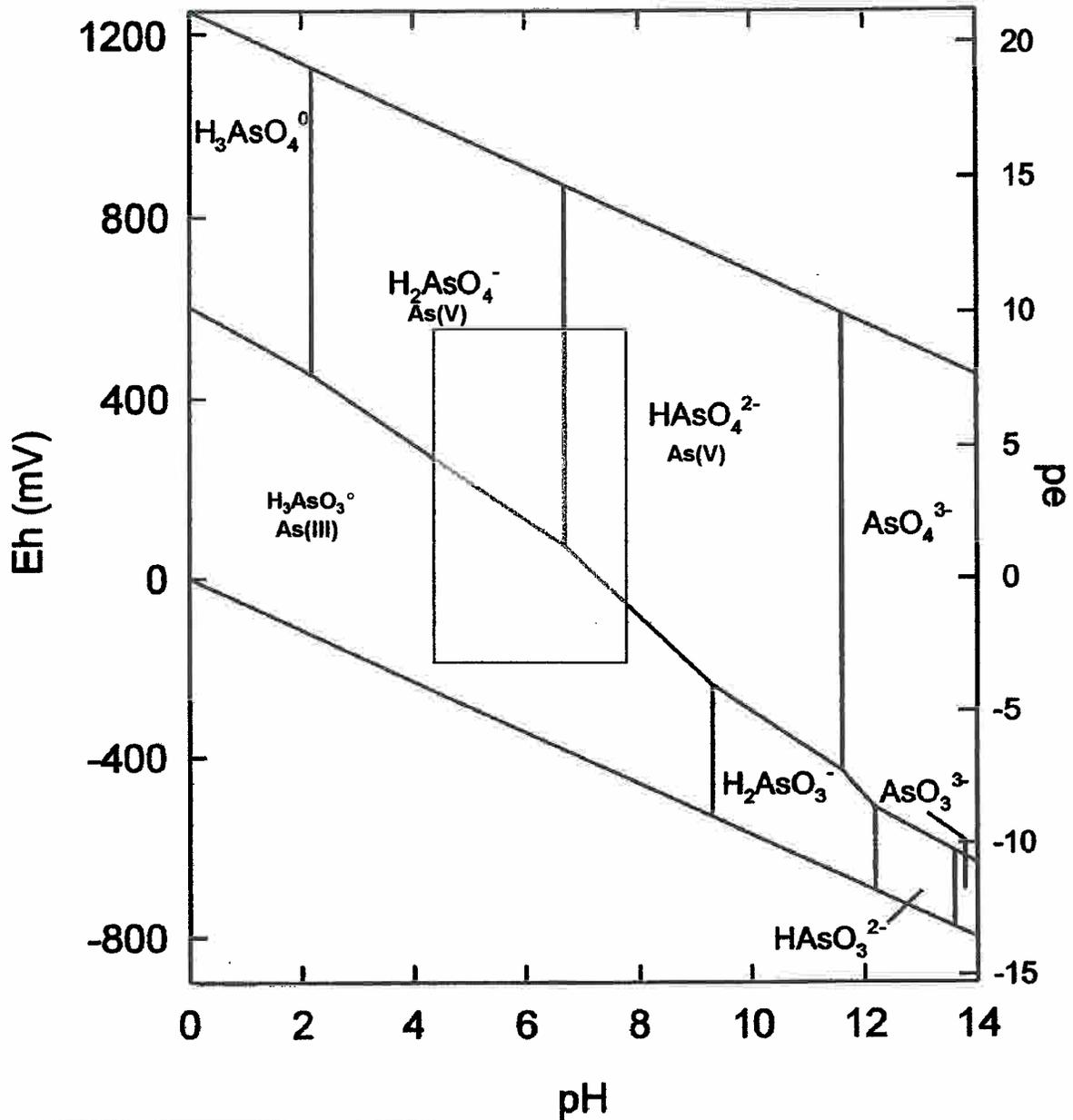
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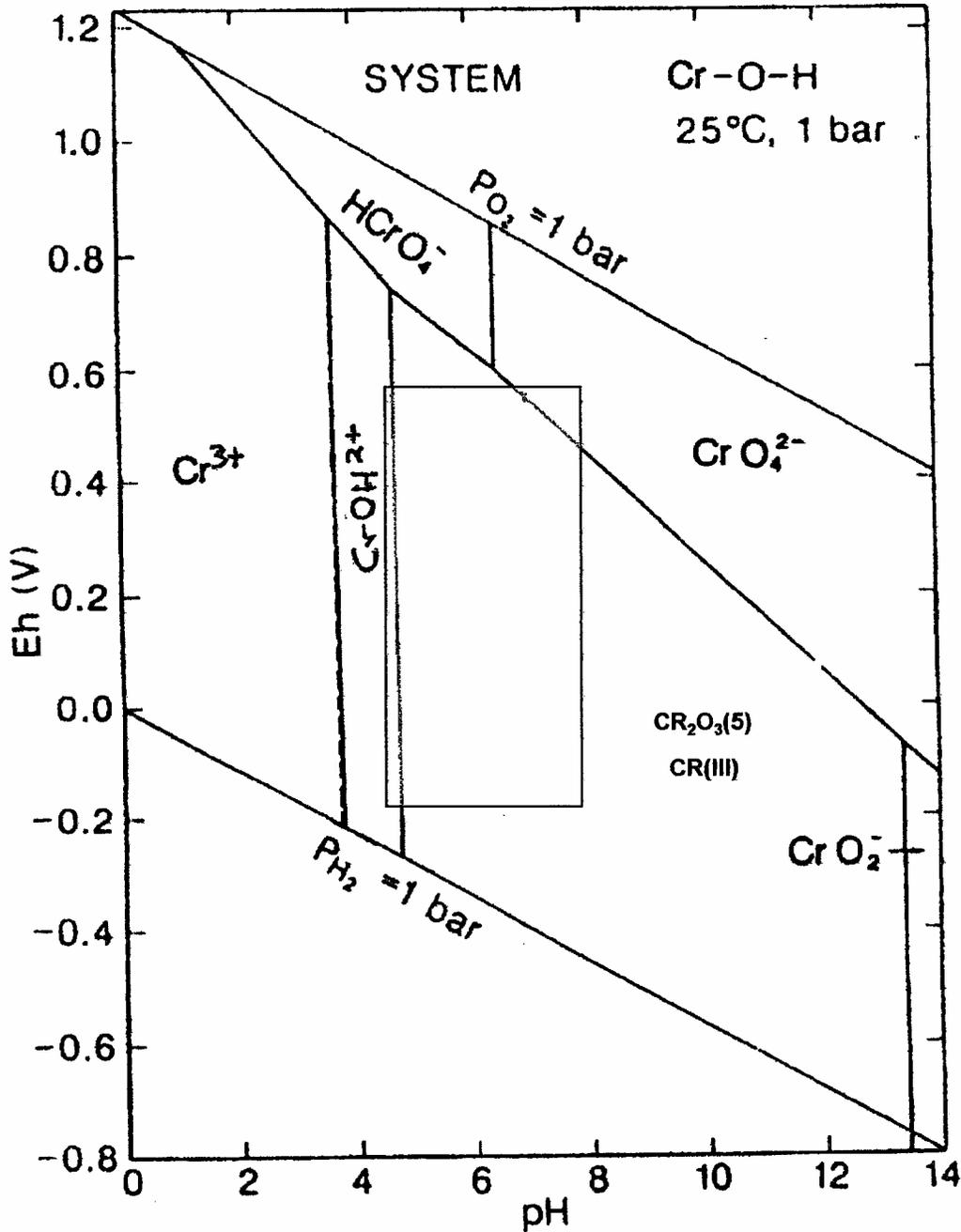


KEY
 □ Site Ground Water Eh-pH Regime

Source: Smedley and Kinniburgh, 2004.

FIGURE 5 SITE Eh-pH SUPERIMPOSED ON Eh-pH DIAGRAM OF AQUEOUS ARSENIC SPECIES IN THE SYSTEM As-O₂-H₂O AT 25°C AND 1 BAR TOTAL PRESSURE WELLS G&H SUPERFUND SITE WOBURN, MASSACHUSETTS		 Boott Mills South Foot of John Street Lowell, MA 01852 978-970-5600
		TRC PROJ. NO.: 02136-0570-01461
		EPA CONTRACT NO.: 68-W6-0042
		RAC SUBCONTRACT NO.: 107061

02136WELLS G&H/OU-3/EH1



KEY
 Site Ground Water Eh-pH Regime

Source: ERG, 2003.

FIGURE 6
SITE Eh-pH REGIME FOR CHROMIUM IN
GROUND WATER SUPERIMPOSED ON Eh-pH DIAGRAM FOR THE
SYSTEM Cr-O₂-H₂O, ASSUMING THAT THE CONCENTRATION
OF Cr=10⁻⁶ mol/kg AT SOLID/LIQUID BOUNDARIES
 WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS



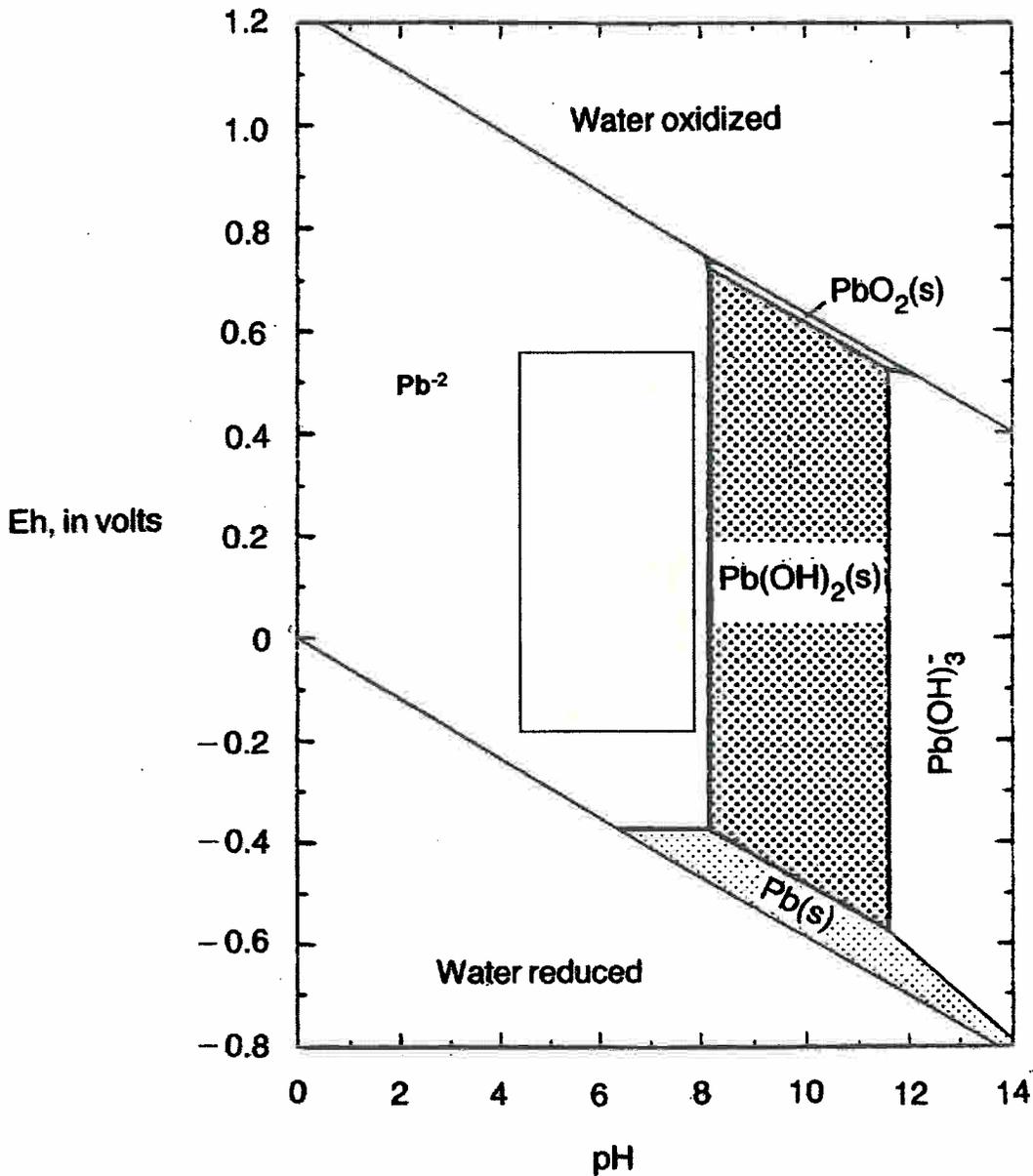
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02136/WELLS G&H/01-3/EH4



At 25°C and 1 atm; lead activity = $10^{-8.3} M$

KEY
 □ Site Ground Water Eh-pH Regime

Source: Bodek et al., 1988.

FIGURE 7
 SITE Eh-pH REGIME FOR LEAD IN
 GROUND WATER SUPERIMPOSED ON Eh-pH
 DIAGRAM FOR THE SYSTEM Pb AND H₂O,

WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS

M&E Metcalf & Eddy

TRC

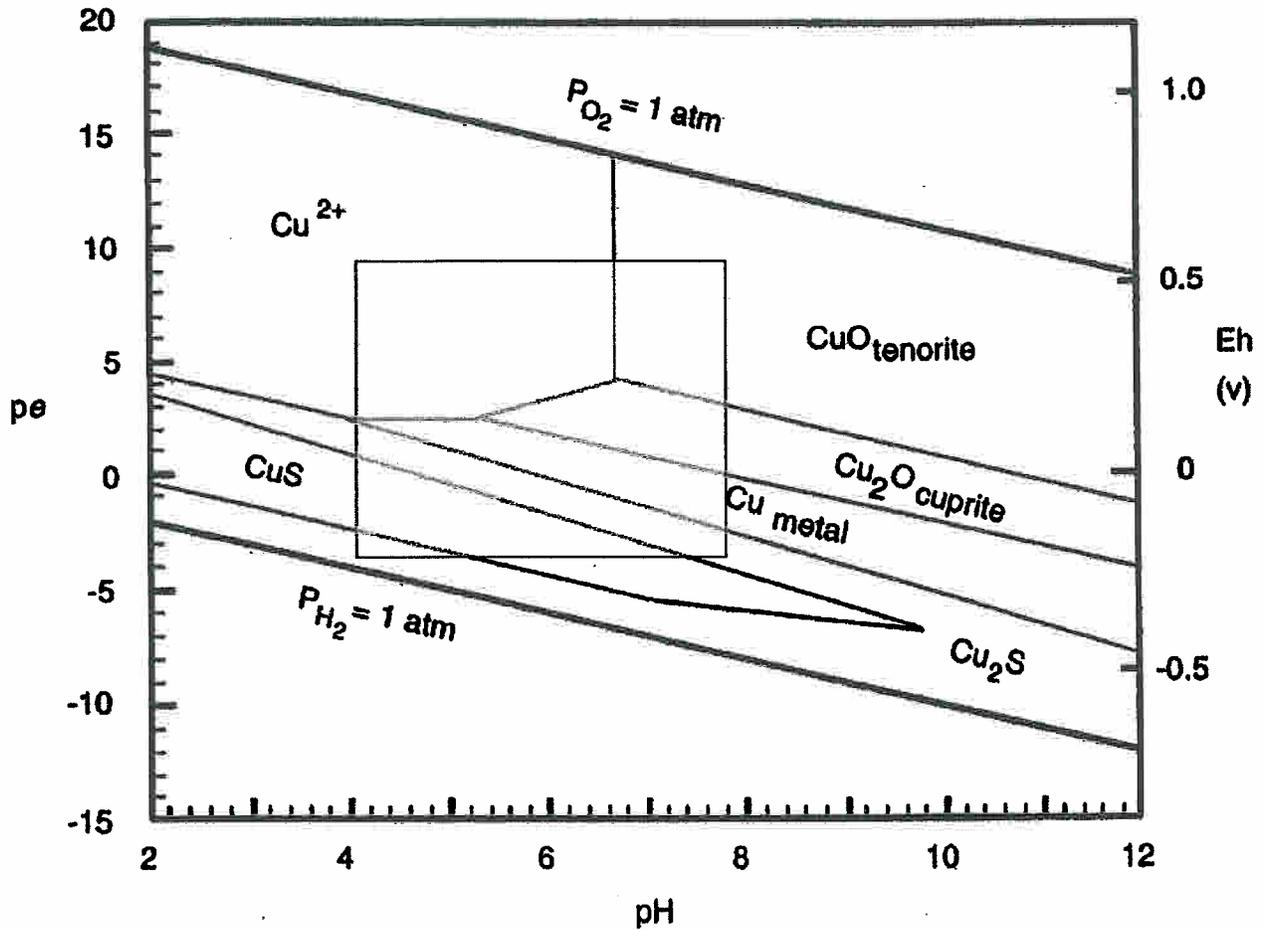
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02136WELLS G&H/OU-3/EMH



KEY

□ Site Ground Water Eh-pH Regime

Source: ERG, 2003.

FIGURE 8
SITE Eh-pH REGIME FOR COPPER IN GROUND WATER
SUPERIMPOSED ON Eh-pH DIAGRAM
FOR THE SYSTEM Cu-O₂-S-H₂O, ASSUMING THAT
 $\Sigma\text{Cu}=10^{-6}$ mol/kg and $\Sigma\text{S}=10^{-2}$ mol/kg
 WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS

M&E Metcalf & Eddy

TRC

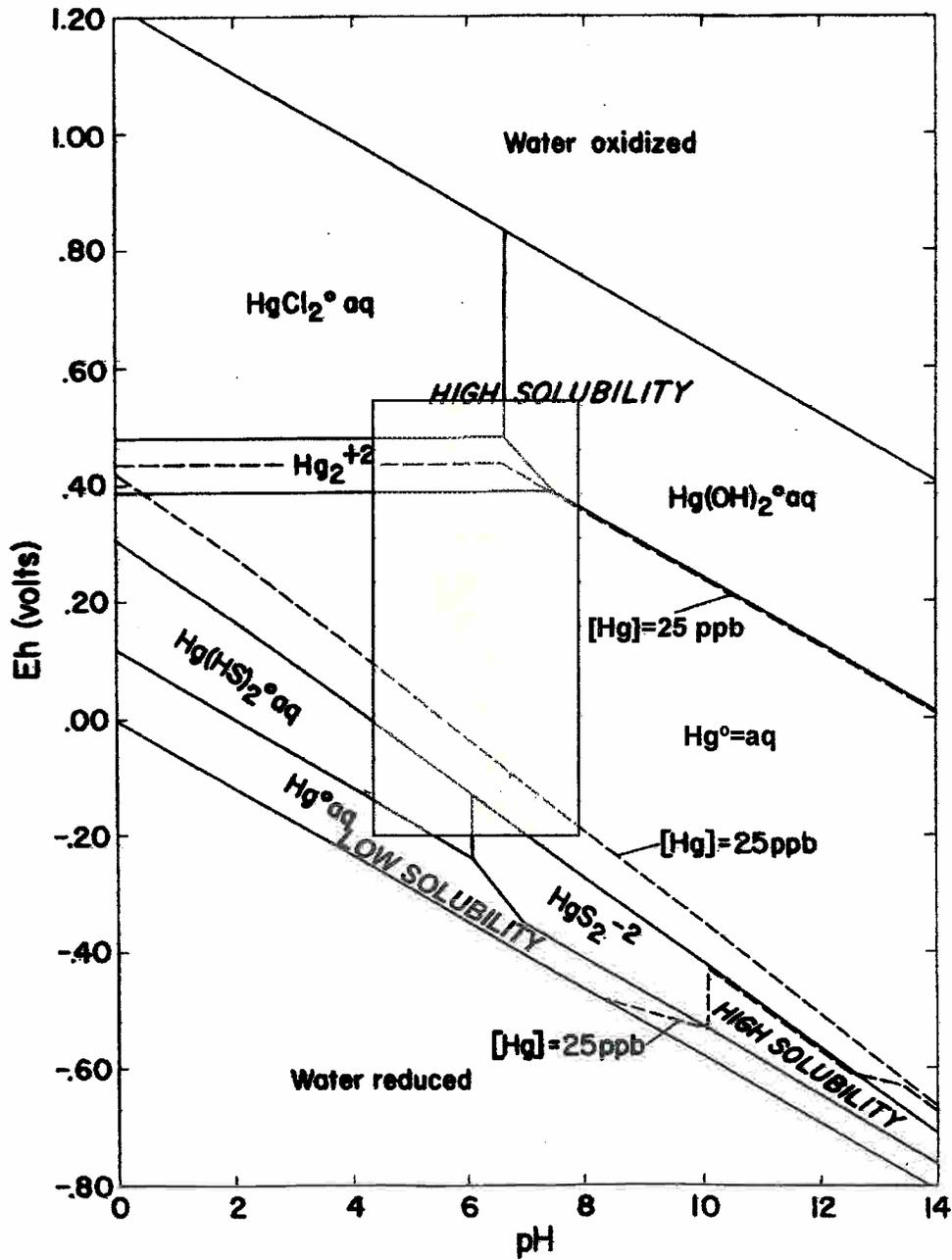
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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061

02136WELLS G&H/PCU-3/EH/5



System contains 36 ppm Cl⁻, 96 ppm sulfur as SO₄⁻².
 Dashed line indicates approximate solubility of mercury in this system.

KEY
 Site Ground Water Eh-pH Regime

Source: Bodek et al., 1988.

FIGURE 9
 SITE Eh-pH REGIME FOR MERCURY IN
 GROUND WATER SUPERIMPOSED ON FIELDS
 OF STABILITY FOR AQUEOUS MERCURY
 SPECIES AT 25°C AND 1 ATMOSPHERE

WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS

M&E Metcalf & Eddy

TRC

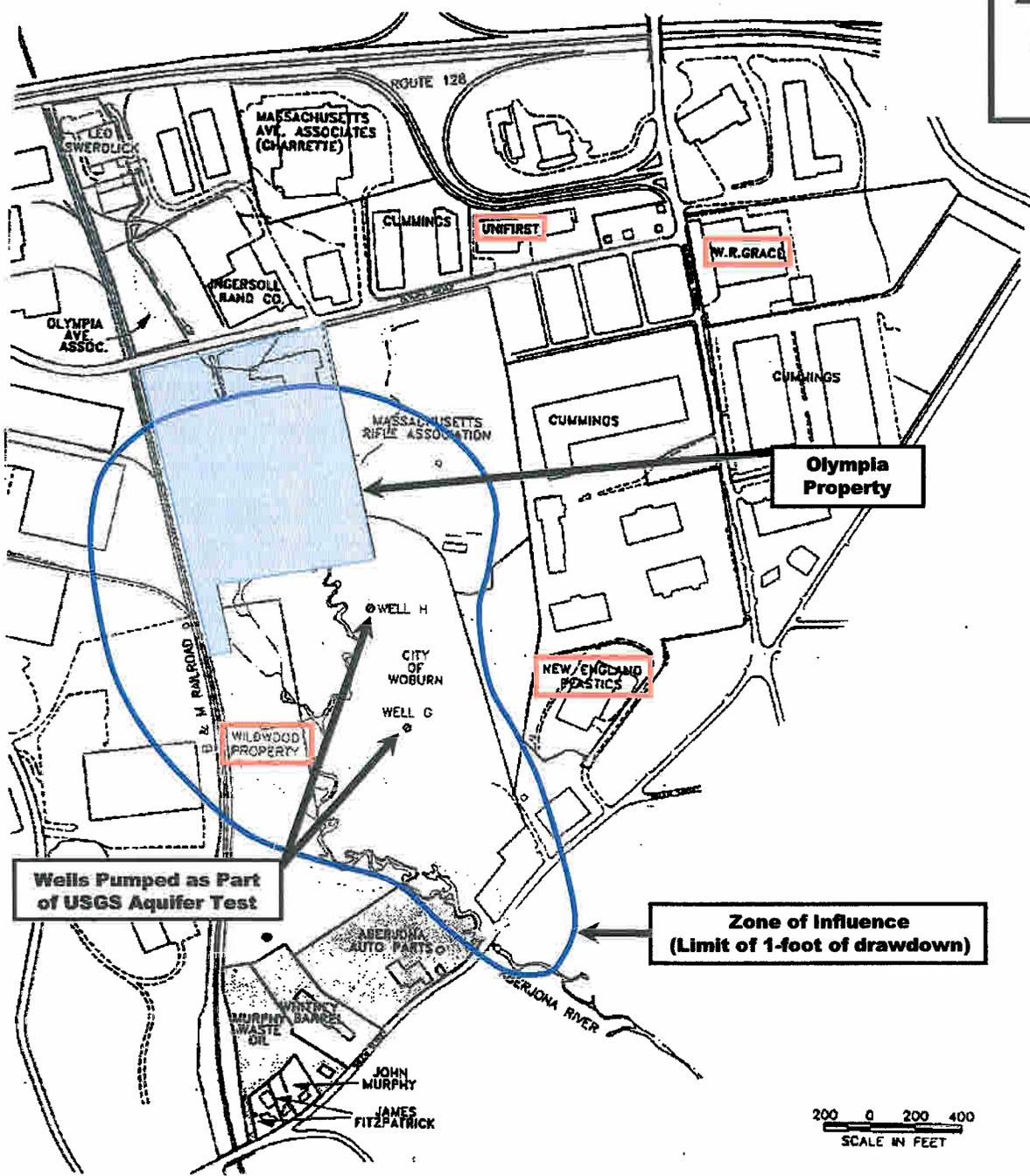
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RAC SUBCONTRACT NO.: 107061

02136WELLS G&H/OU-3/EHT



Zone of Influence reference: USGS, 1987. Myette, C.F., Olimpio, J.C., and Johnson, D.G. Area of Influence and Zone of Contribution to Superfund-Site Wells G&H, Woburn, Massachusetts: U.S. Geological Survey WRI 87-4100. 1987

LEGEND

Other Wells G&H Source Areas

FIGURE 10
WELLS G&H ZONE OF INFLUENCE –
DRAWDOWN OF WATER TABLE
AFTER 30 DAY PUMPAGE,
JANUARY 3, 1986
 WELLS G&H OU1
 OLYMPIA PROPERTY SUPERFUND SITE
 WOBURN, MASSACHUSETTS

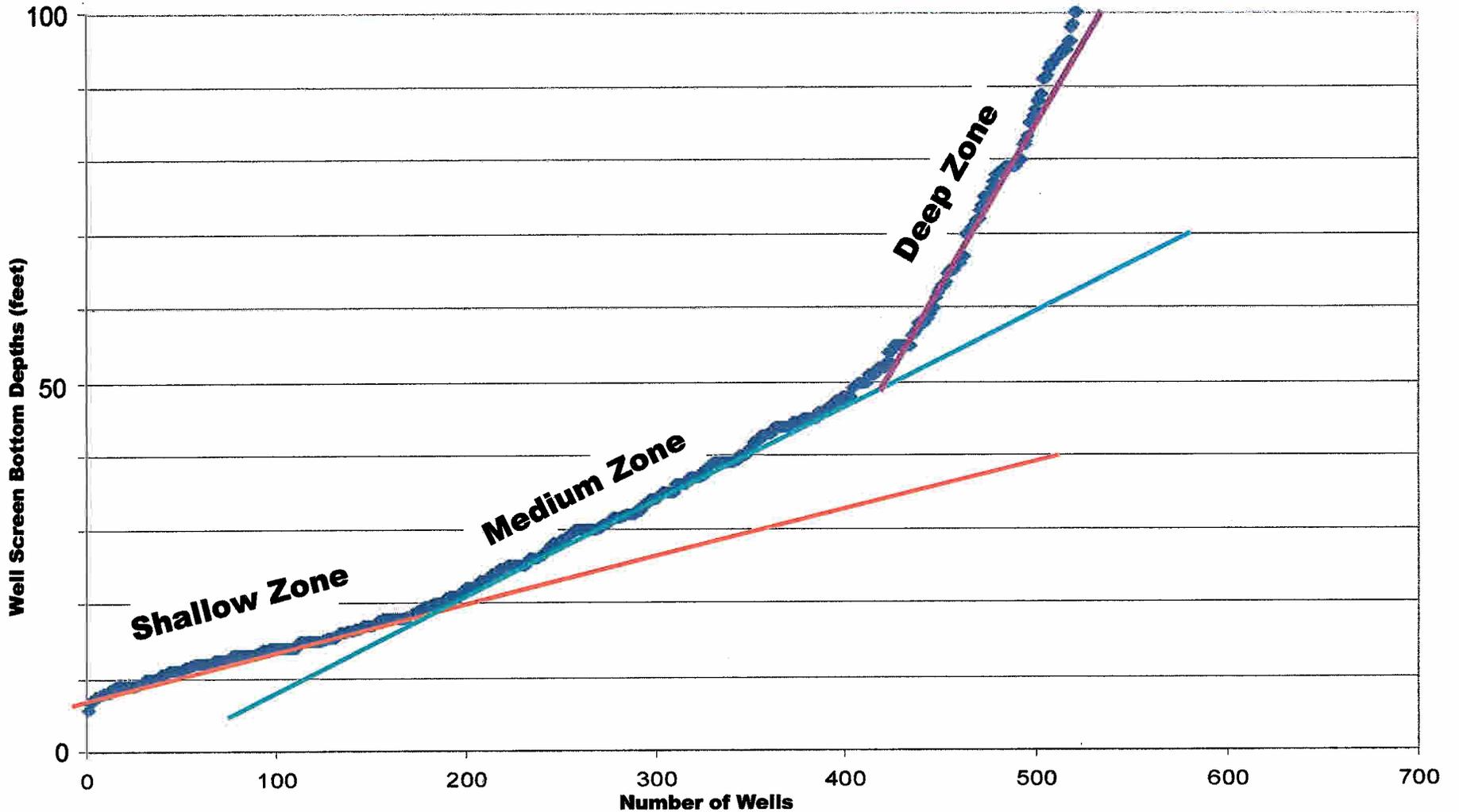


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QUADRANGLE LOCATION
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EPA CONTRACT NO.: 68-W6-0042
RAC SUBCONTRACT NO.: 107061





02136/WELLS G&H/OU-3/CHART

FIGURE 11
MONITORING WELL DEPTH ZONE SELECTION
PLOT OF GROUNDWATER WELL SCREEN BOTTOM
DEPTHS BY FREQUENCY DENOTING NATURAL BREAKS
IN NUMBER OF WELLS BY BOTTOM SCREEN DEPTH
 WELLS G&H
 SUPERFUND SITE
 WOBURN, MASSACHUSETTS



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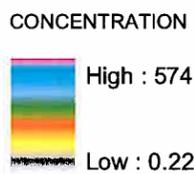
TRC PROJ. NO.: 02136-0570-01461

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 10 ug/L for Arsenic

Date Range
1987 - 2002

Figure 12A
0 - 20 Foot Arsenic
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA



TRC Boot Mills South
Foot of John Street
Lowell, MA 01852
(978)970-5600

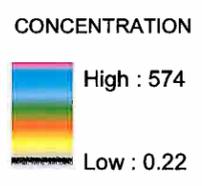
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 10 ug/L for Arsenic

Date Range
1980 - 2002



Figure 12B
>20 - 50 Foot Arsenic
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

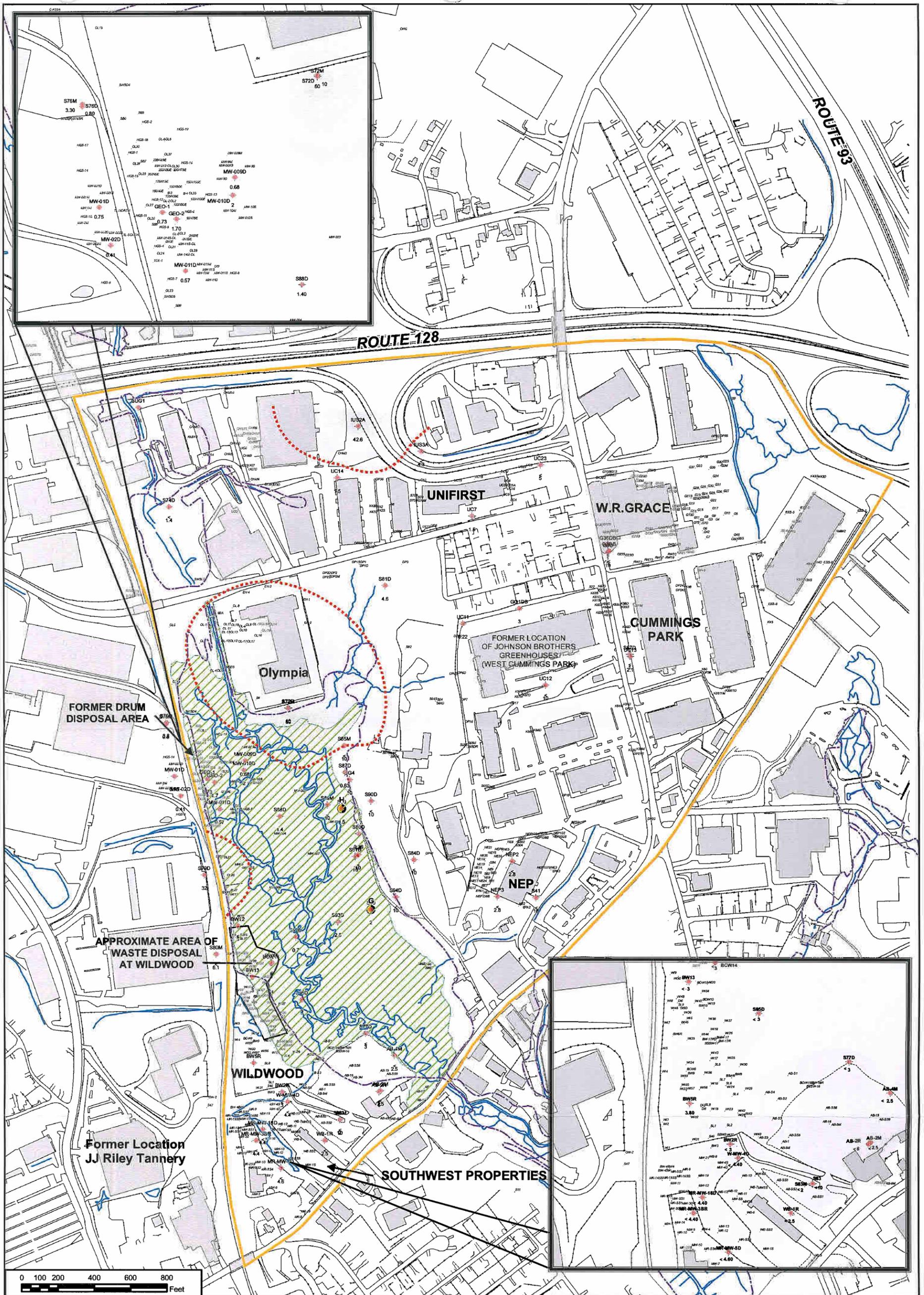


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(978)970-5600

TRC PROJ. NO.: 02136-0570-0146H

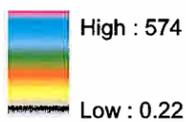
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 10 ug/L for Arsenic

Date Range
1979 - 2002



Figure 12C
>50 Foot Arsenic
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

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Lowell, MA 01852
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TRC PROJ. NO.: 02136-0570-0146H

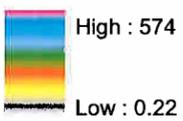
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 10 ug/L for Arsenic

Date Range
2001 - 2002



Figure 12D
0 - 20 Foot Arsenic
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

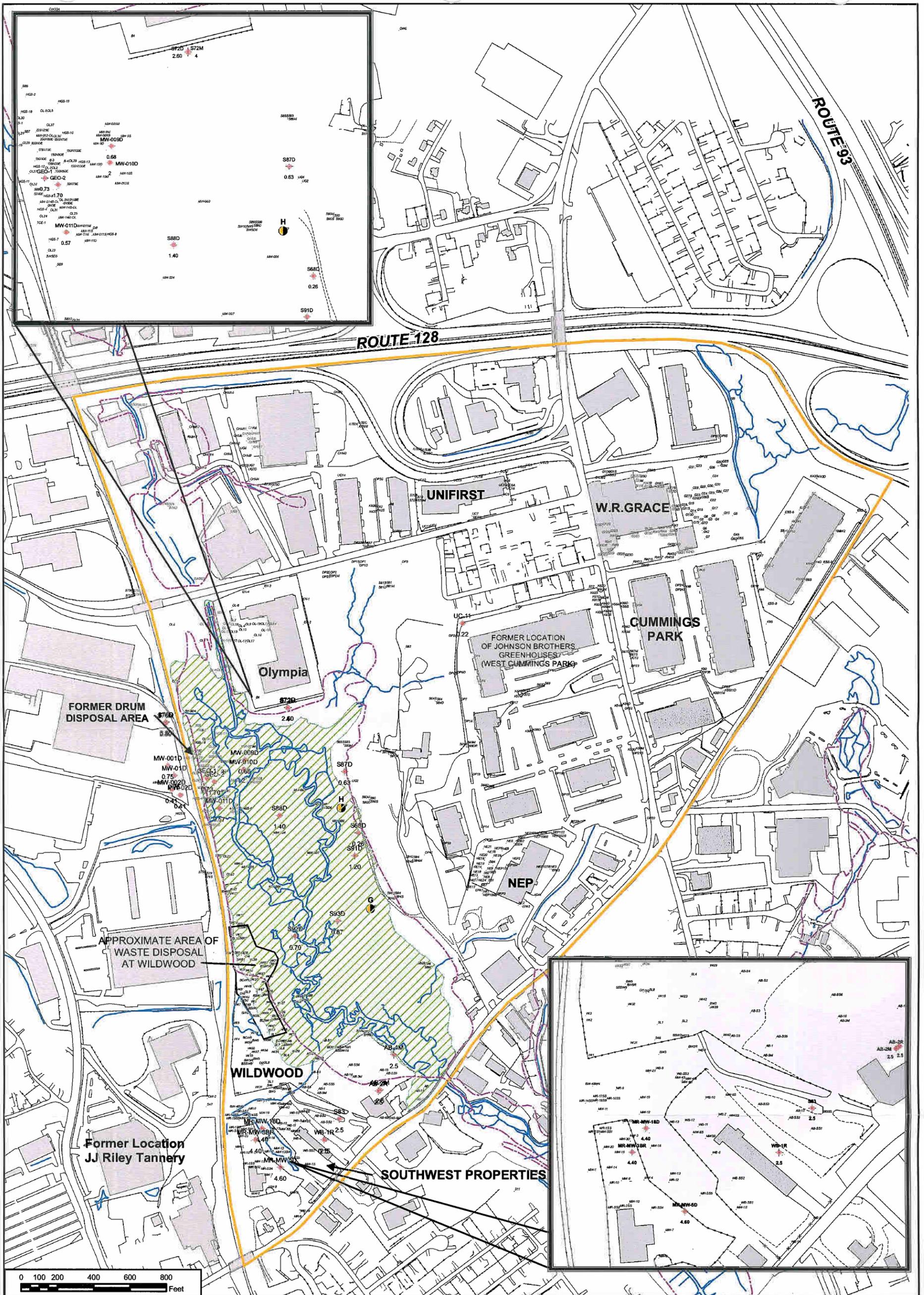
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Foot of John Street
Lowell, MA 01852
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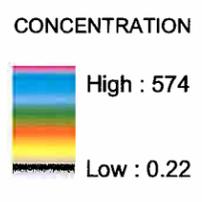
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 10 ug/L for Arsenic

Date Range
2001 - 2002



Figure 12F
>50 Foot Arsenic
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA



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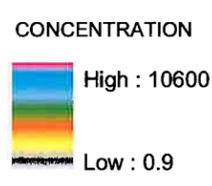
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 100 ug/L for Total Chromium

Date Range
1980 - 2002

Figure 13A
0 - 20 Foot Chromium
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

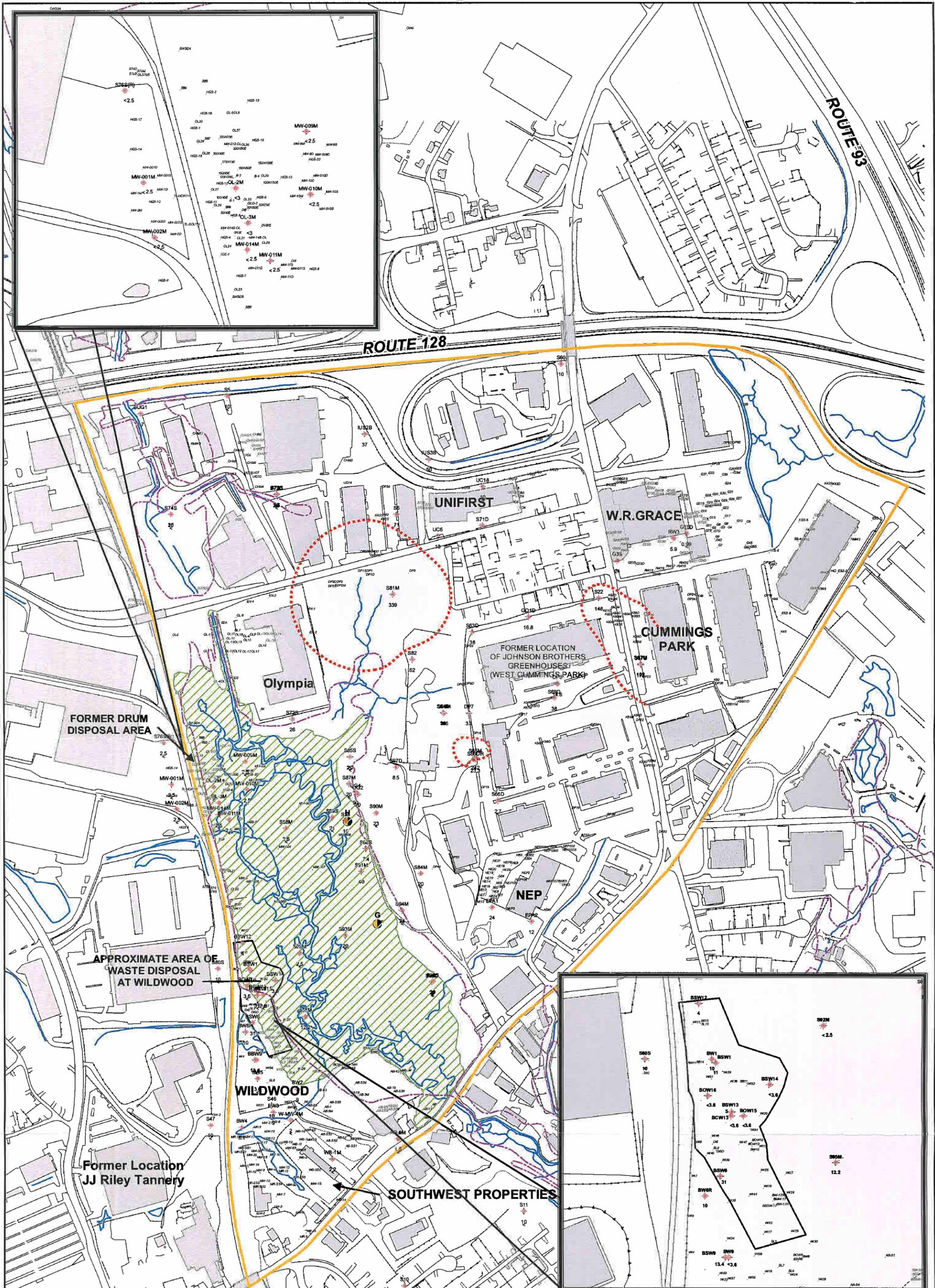


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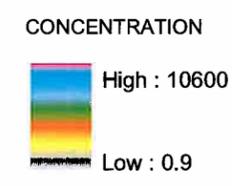
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 100 ug/L for Total Chromium

Date Range
1980 - 2002



Figure 13B
>20 - 50 Foot Chromium
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

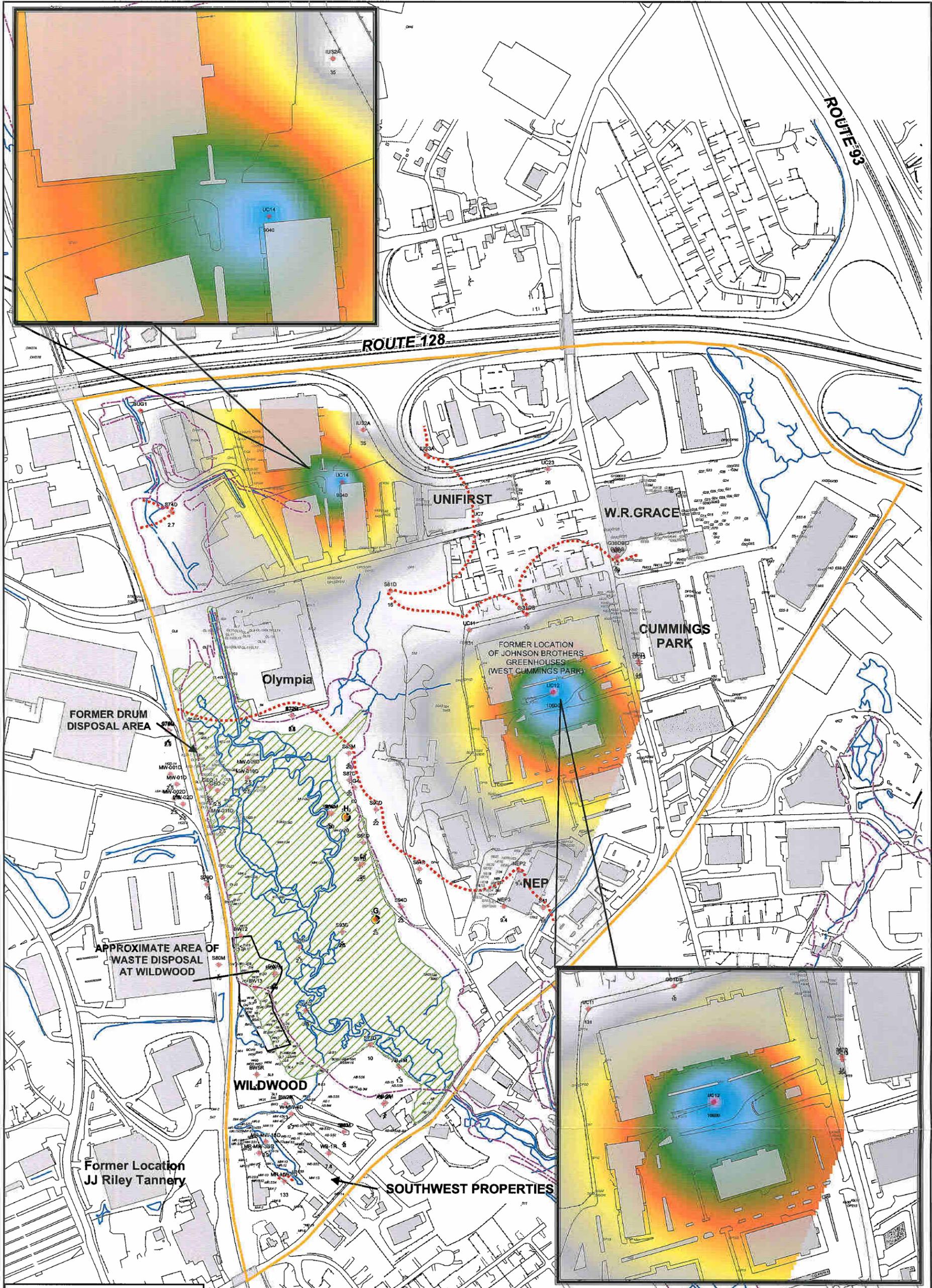


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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



Date Range
1981- 2002

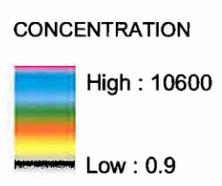
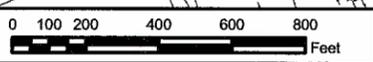
Figure 13C
>50 Foot Chromium
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

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Foot of John Street
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TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061

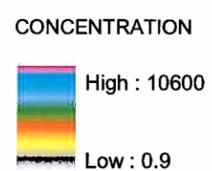


- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 100 ug/L for Total Chromium





All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 100 ug/L for Total Chromium

Date Range
2001 - 2002

Figure 13D
0 - 20 Foot Chromium
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

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TRC PROJ. NO.: 02136-0570-0146H

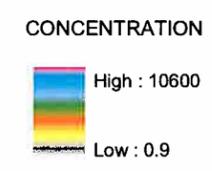
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061





All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA MCL of 100 ug/L for Total Chromium

Date Range
2001 - 2002



Figure 13E
>20 - 50 Foot Chromium
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

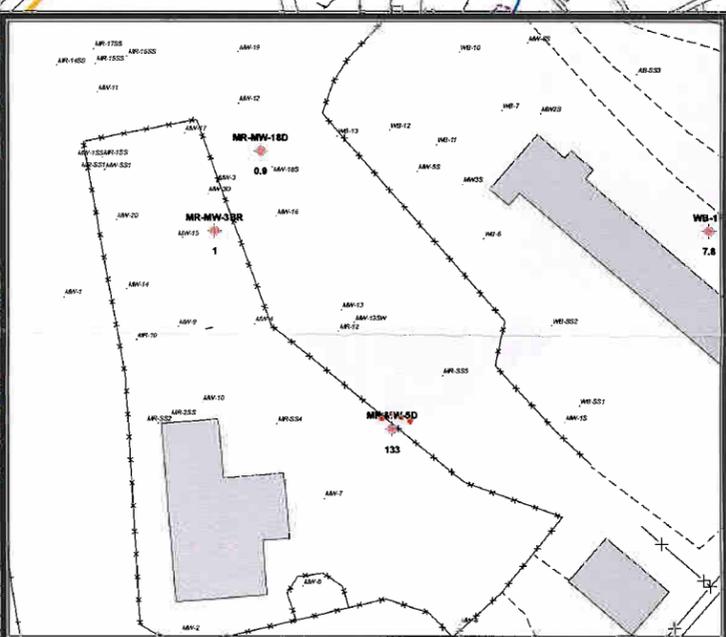
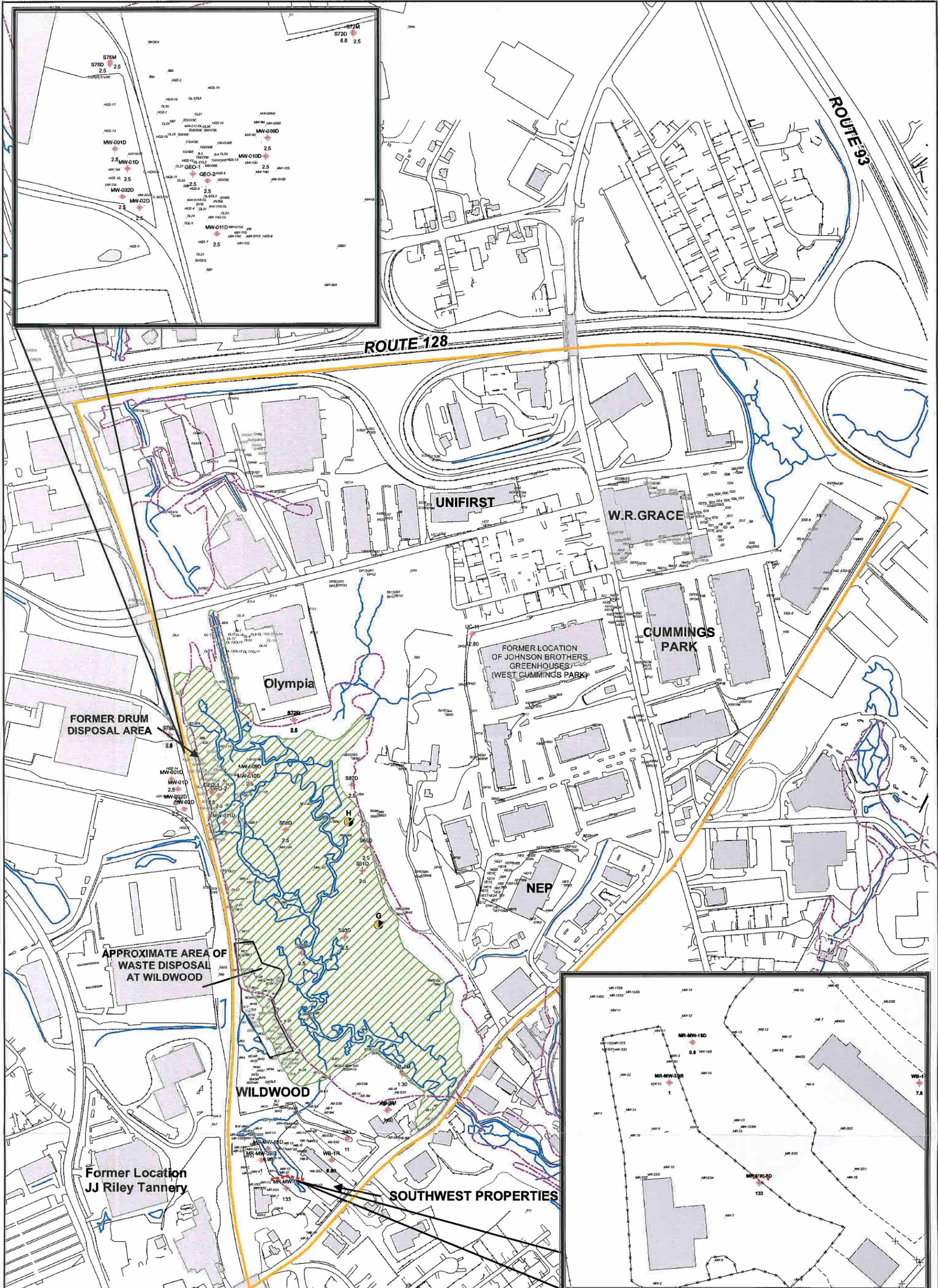


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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



0 100 200 400 600 800 Feet

<p>CONCENTRATION</p> <p>High : 10600</p> <p>Low : 0.9</p>	<p>INORGANIC SAMPLE SITES</p>	<p>Date Range 2001- 2002</p>
	<p>SAMPLE SITES</p>	<p>WELLS G&H SITE BOUNDARY</p>
	<p>WETLAND EXTENT</p>	<p>WELLS G&H</p>
	<p>100 YEAR FLOODPLAIN</p>	<p>Areas in excess of EPA MCL of 100 ug/L for Total Chromium</p>
	<p>N</p>	

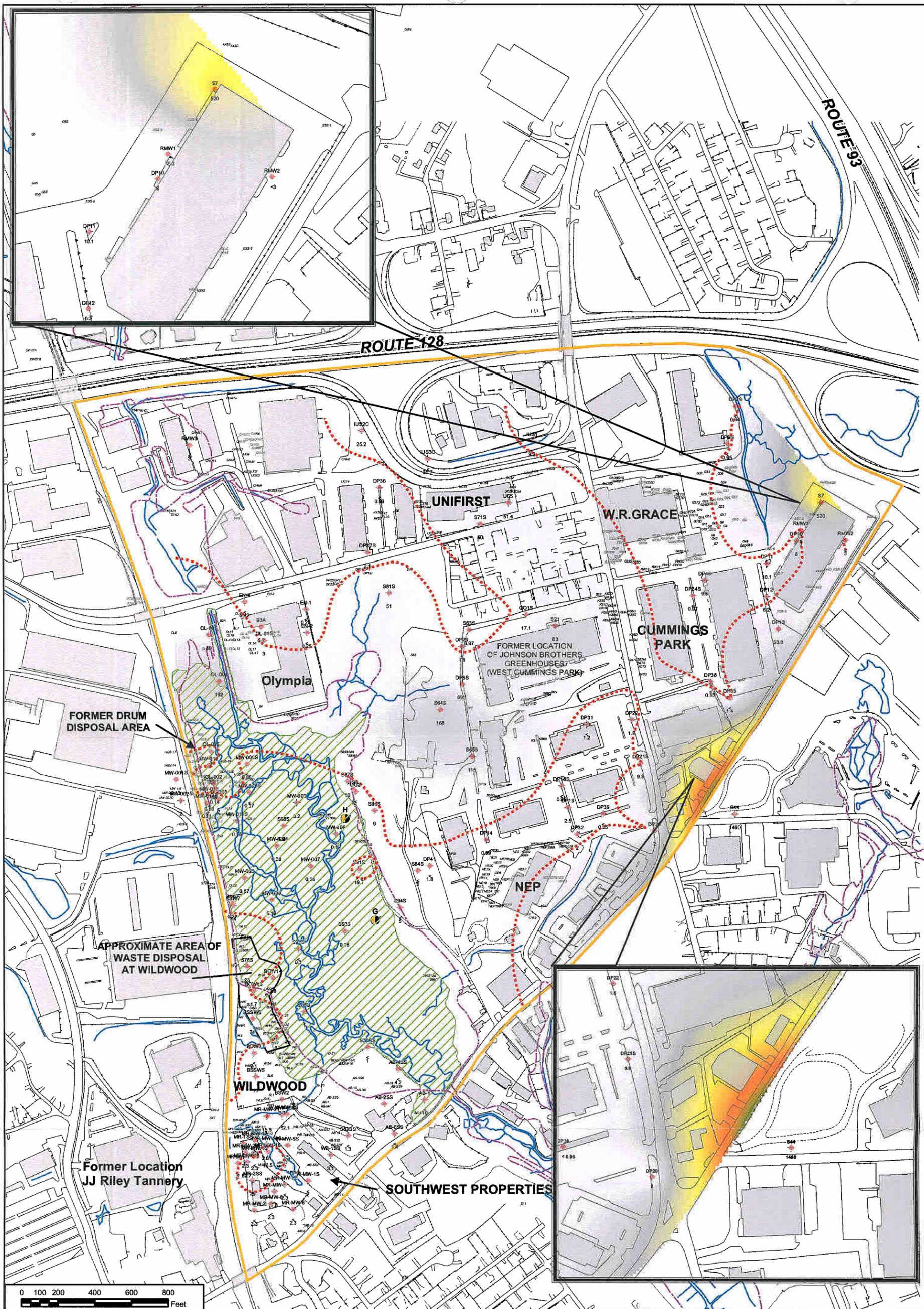
Figure 13F
>50 Foot Chromium
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

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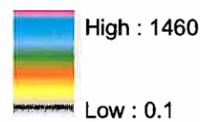
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA of 15 ug/L Action Level for Lead

Date Range
1980- 2002



Figure 14A
0 - 20 Foot Lead
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

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Lowell, MA 01852
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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

- | | | | |
|---------------|--|-------------|-----------|
| CONCENTRATION | | High : 1460 | Low : 0.1 |
| | INORGANIC SAMPLE SITES | | |
| | SAMPLE SITES | | |
| | WELLS G&H SITE BOUNDARY | | |
| | WETLAND EXTENT | | |
| | WELLS G&H | | |
| | 100 YEAR FLOOD PLAIN | | |
| | Areas in excess of EPA 15 ug/L Action Level for Lead | | |

Date Range
2001 - 2002

Figure 14D
0 - 20 Foot Lead
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

TRC Boott Mills South
Foot of John Street
Lowell, MA 01852
(978)970-5600

TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061

M&E Metcalf & Eddy



0 100 200 400 600 800 Feet

All plotted data are maximum values

CONCENTRATION

High : 1460
Low : 0.1

- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA 15 ug/L Action Level for Lead

Date Range
1981- 2002



Figure 14C
>50 Foot Lead
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

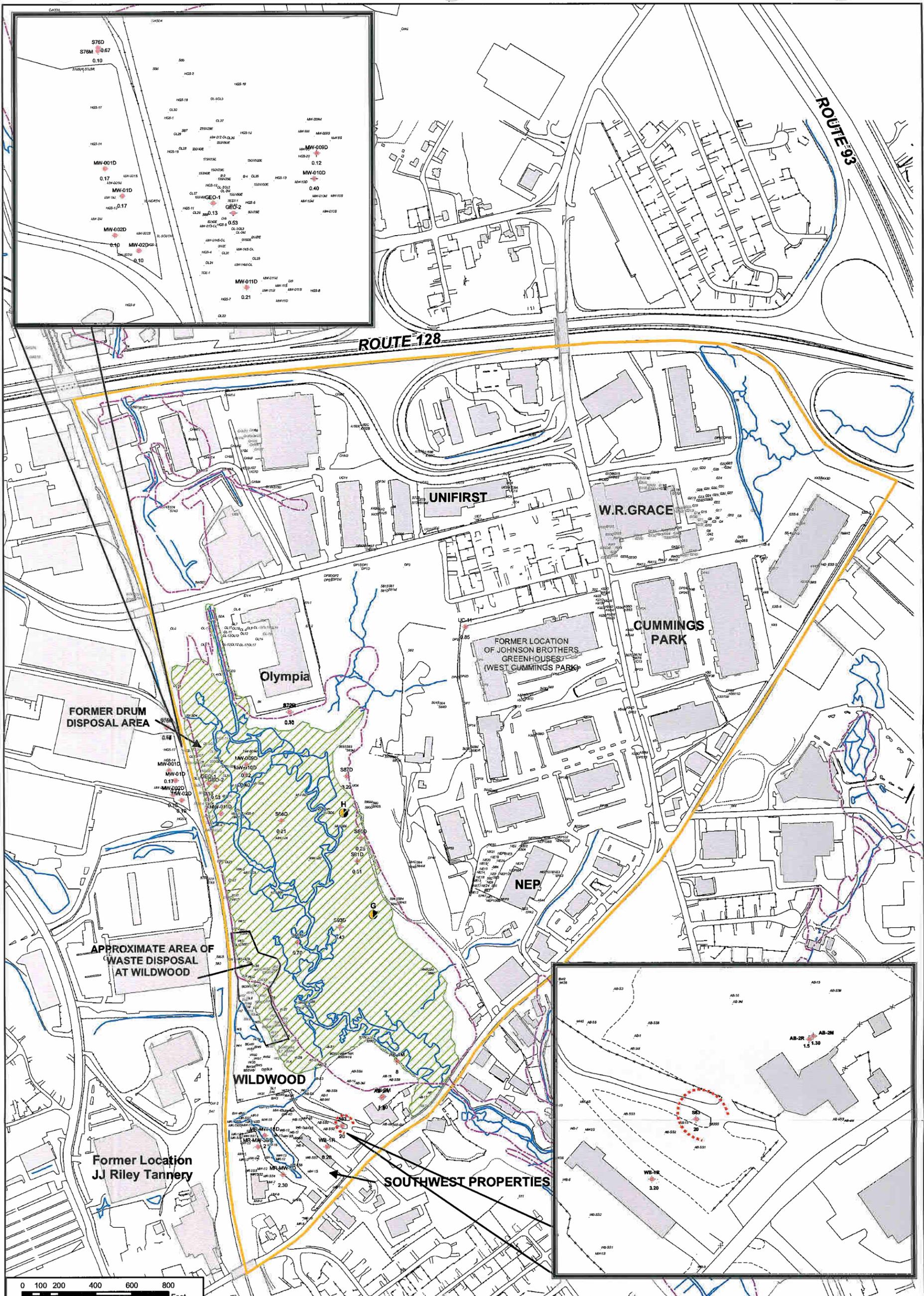


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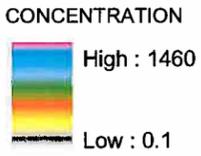
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN
- Areas in excess of EPA 15 ug/L Action Level for Lead

Date Range
2001 - 2002



Figure 14F
>50 Foot Lead
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

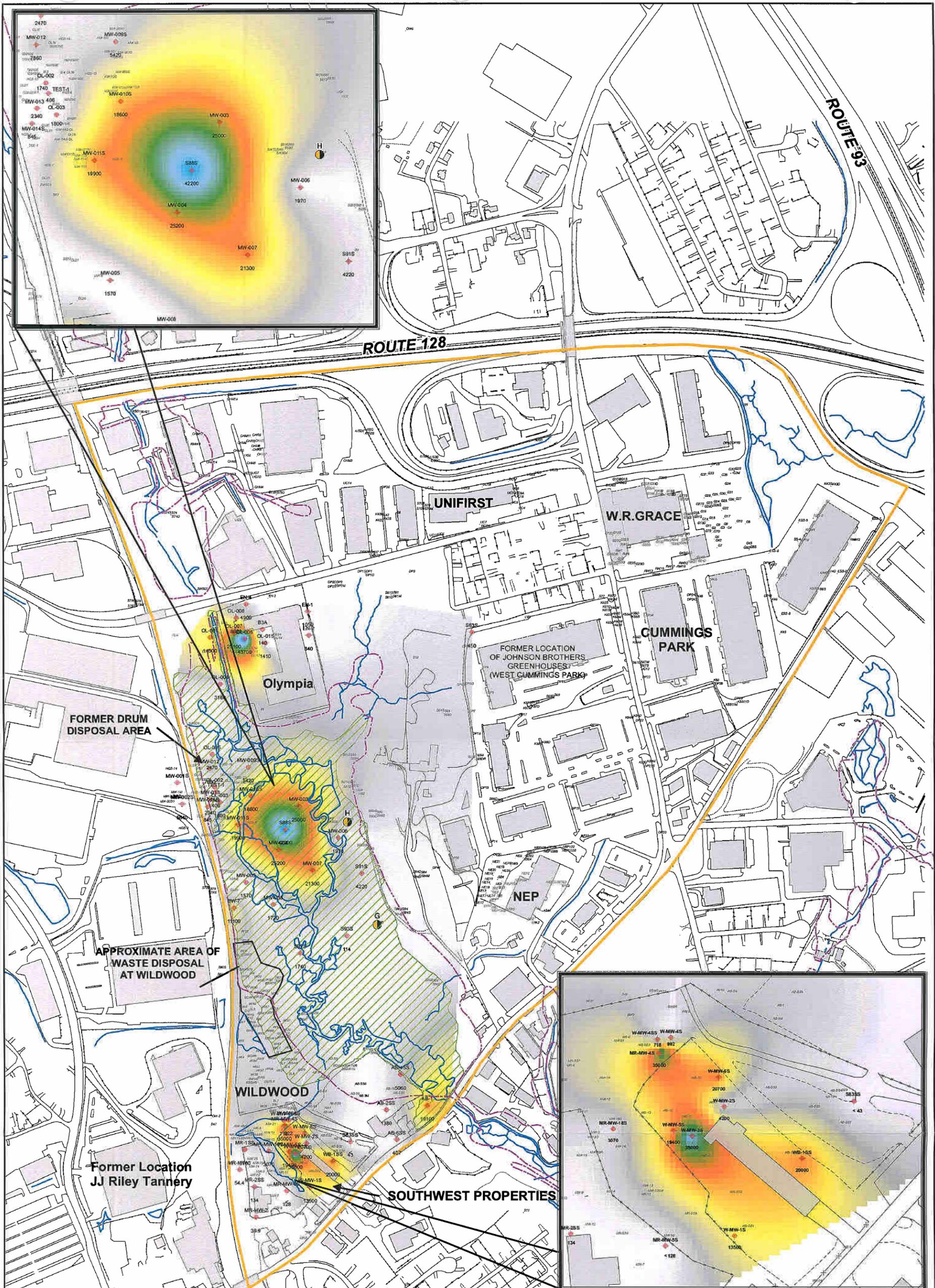


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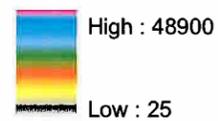
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1993 - 2002



Figure 15A
0 - 20 Foot Iron
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

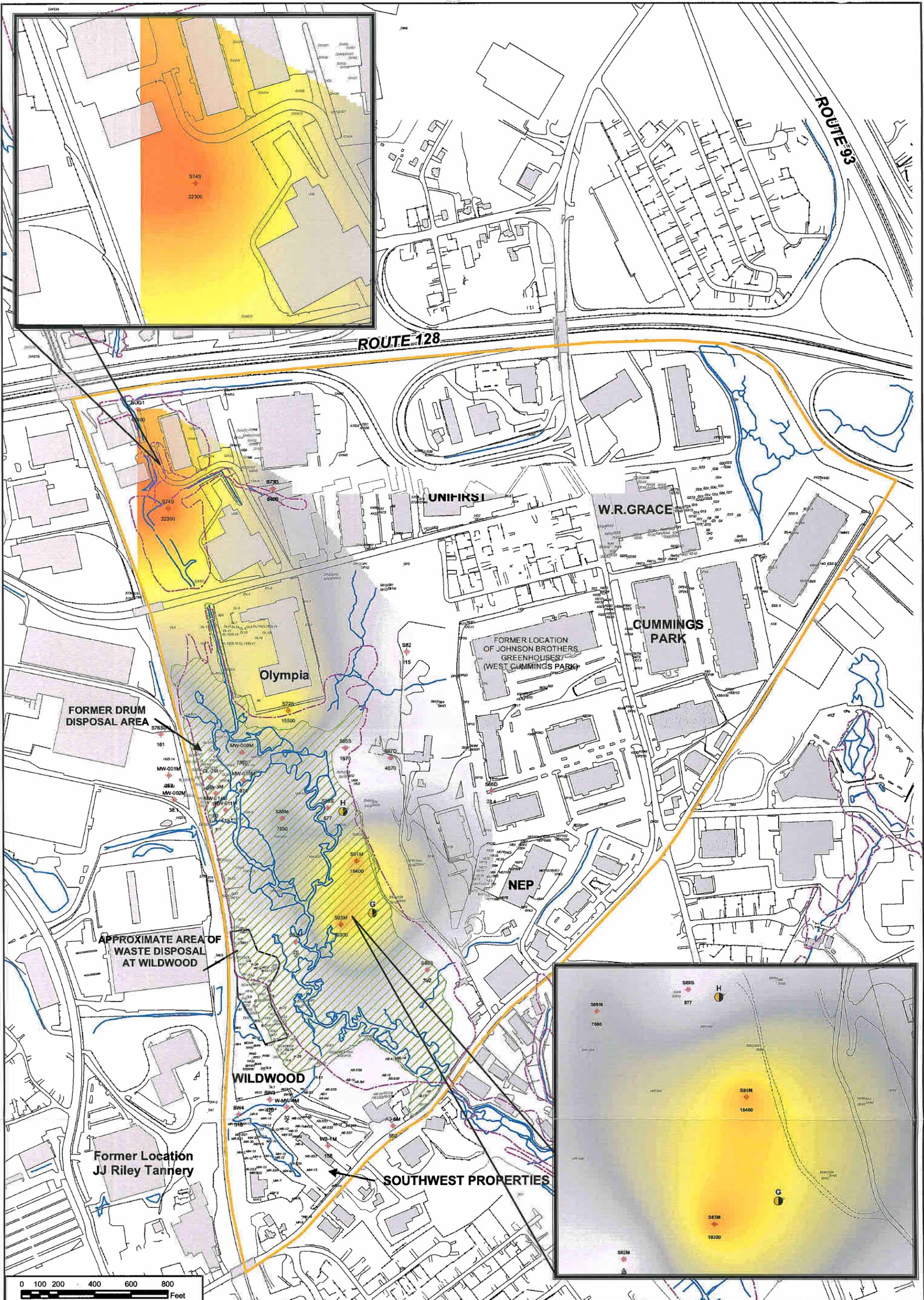
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Lowell, MA 01852
(978)970-5600

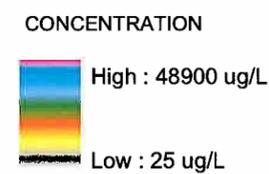
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- ⊙ SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOOD PLAIN

Date Range
1993 - 2002



Figure 15B
>20 - 50 Foot Iron
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

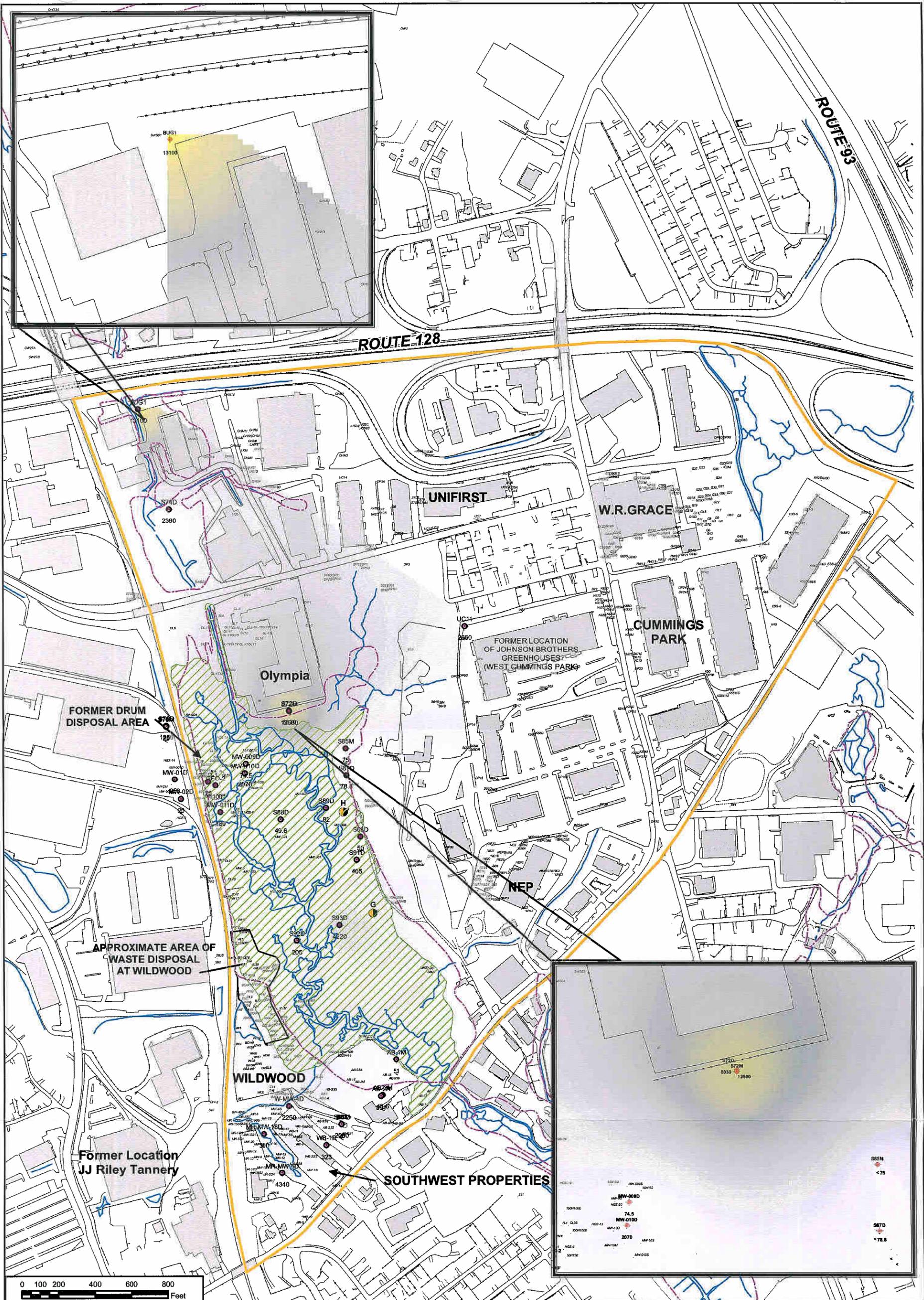


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TRC PROJ. NO.: 02136-0570-0146H

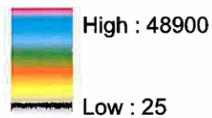
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1985 - 2002



Figure 15C
>50 Foot Iron
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

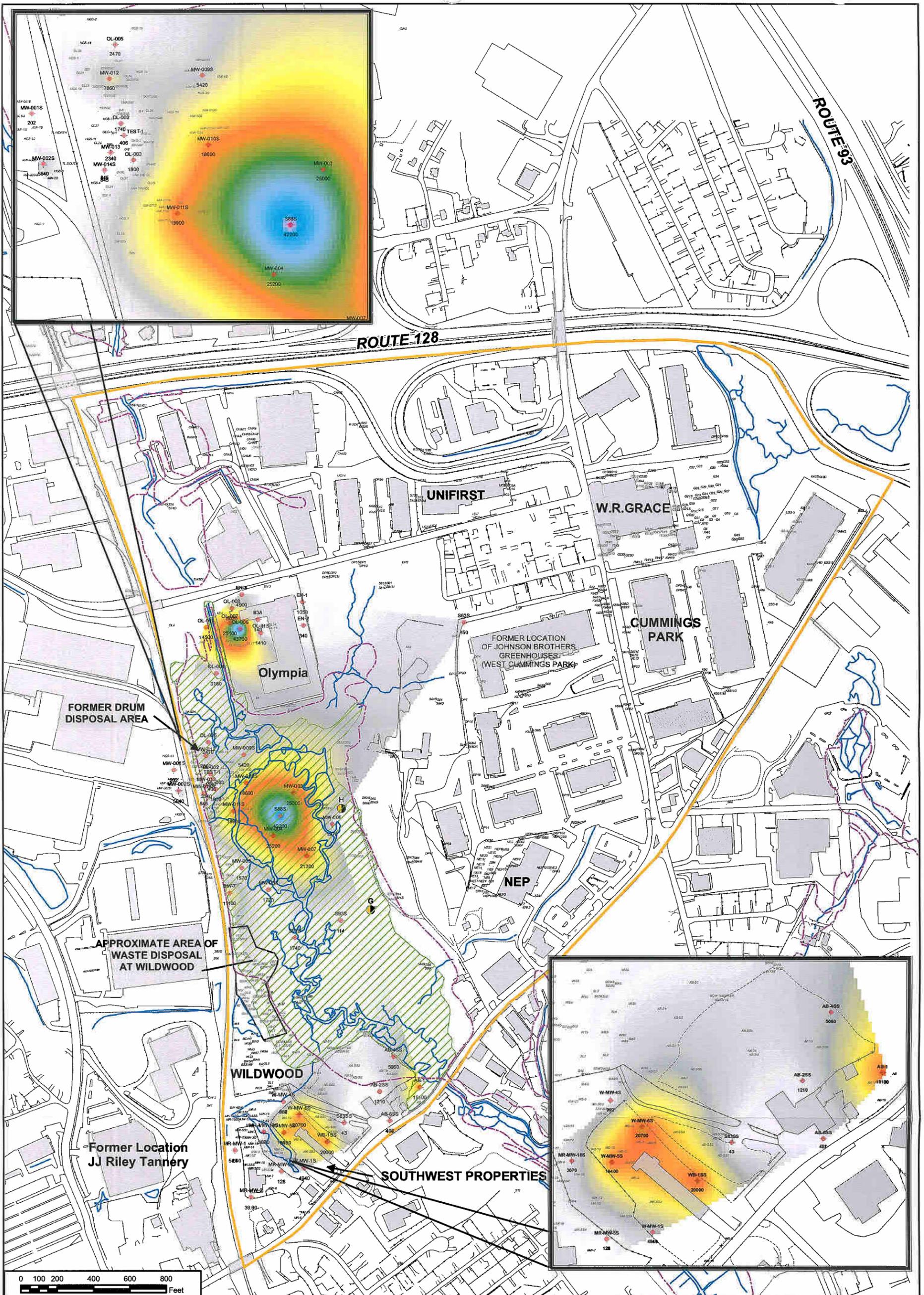
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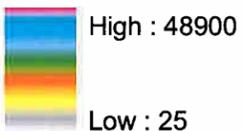
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- - - 100 YEAR FLOODPLAIN

Date Range
2001 - 2002



Figure 15D
0 - 20 Foot Iron
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

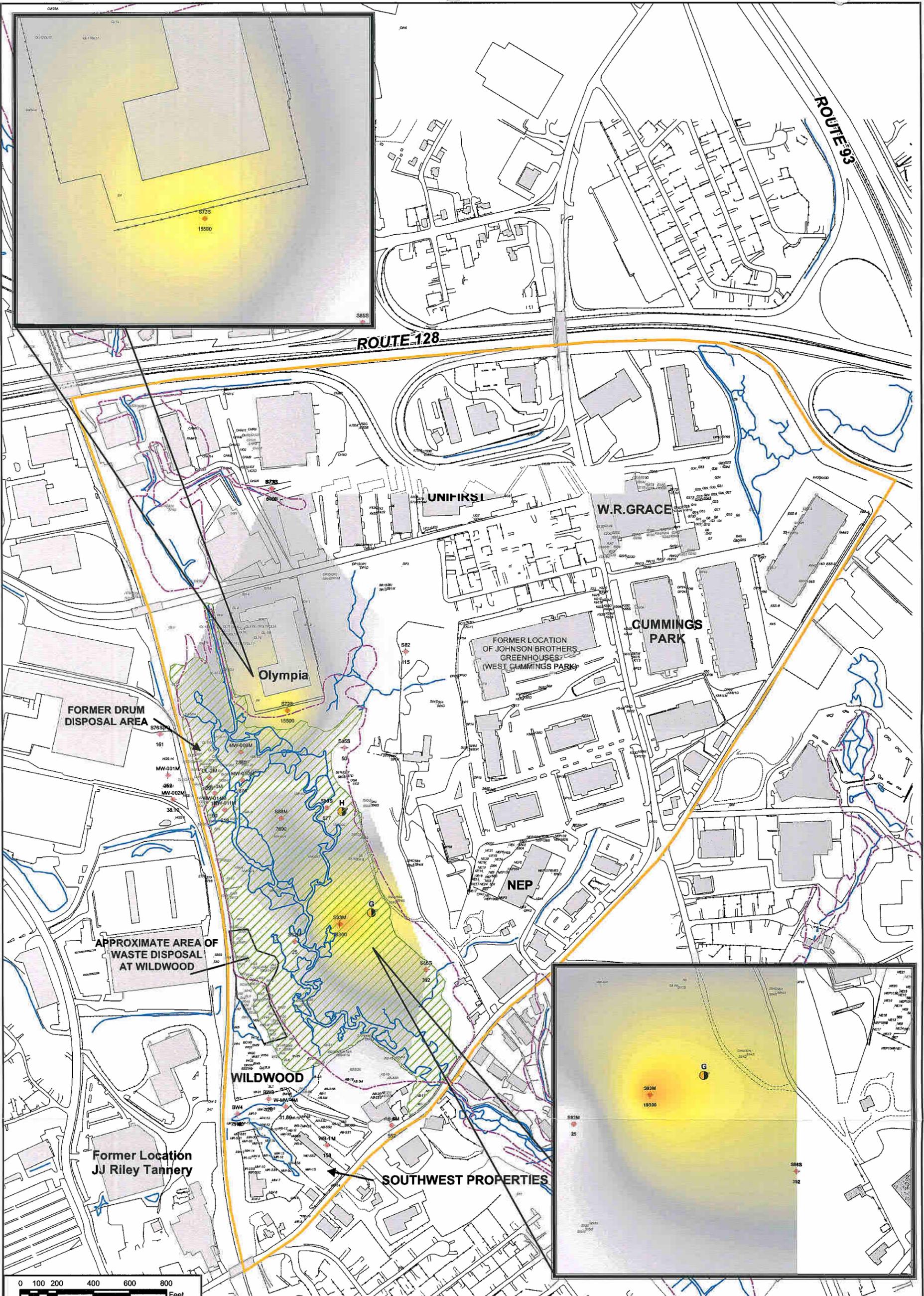
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TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

- | | |
|---------------|---------------------------|
| CONCENTRATION | ◆ INORGANIC SAMPLE SITES |
| High : 48900 | ⊙ SAMPLE SITES |
| Low : 25 | ▭ WELLS G&H SITE BOUNDARY |
| | ▨ WETLAND EXTENT |
| | ● WELLS G&H |
| | — 100 YEAR FLOODPLAIN |

Date Range
2001 - 2002



Figure 15E
>20 - 50 Foot Iron
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

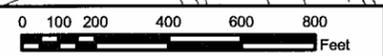
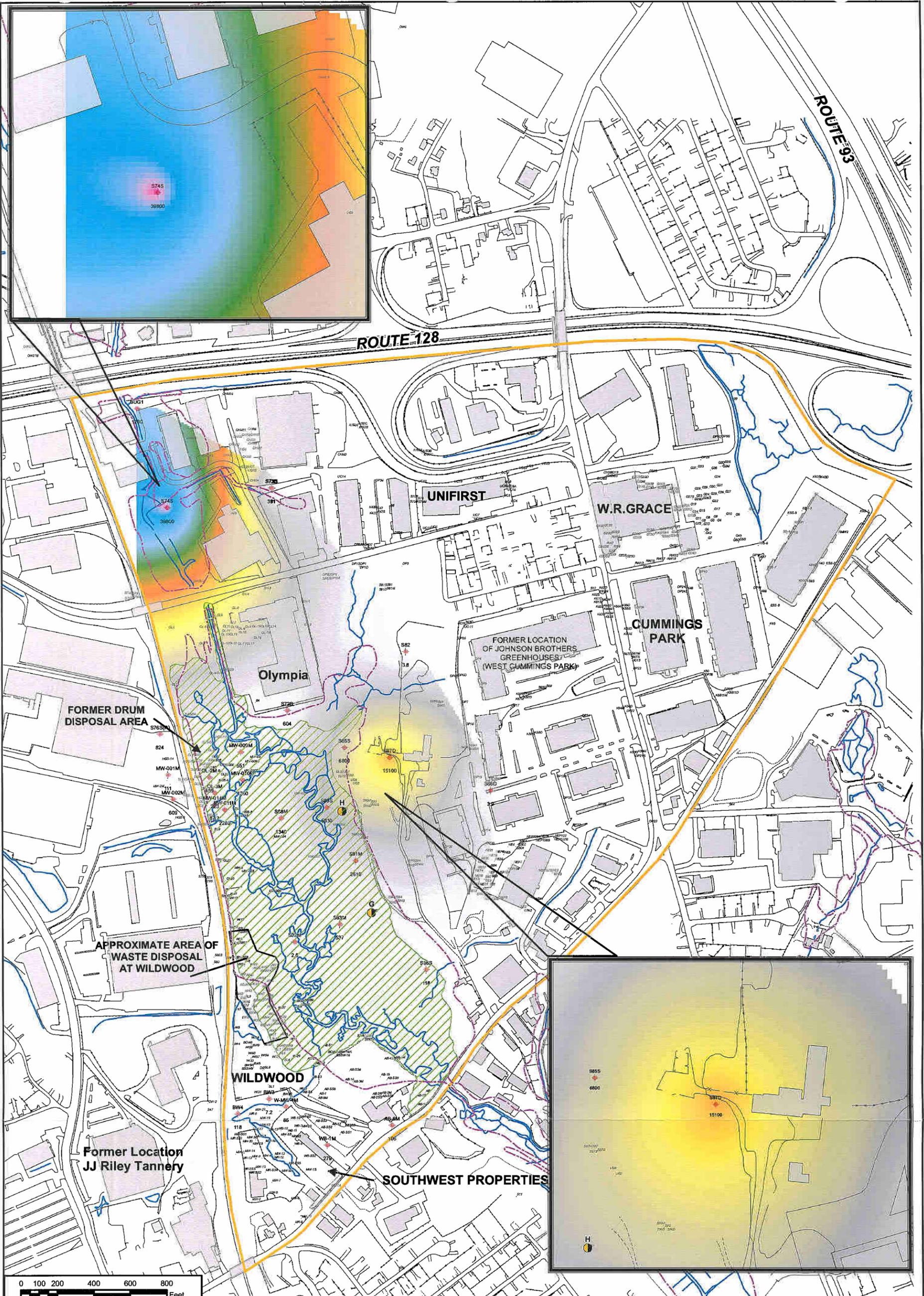


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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

- | | |
|-------------------|---------------------------|
| CONCENTRATION | ◆ INORGANIC SAMPLE SITES |
| High : 39800 ug/L | ⊙ SAMPLE SITES |
| Low : 2.8 ug/L | ▭ WELLS G&H SITE BOUNDARY |
| | ▨ WETLAND EXTENT |
| | ● WELLS G&H |
| | — 100 YEAR FLOODPLAIN |

Date Range
1993 - 2002



Figure 16B
20 - 50 Foot Manganese
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

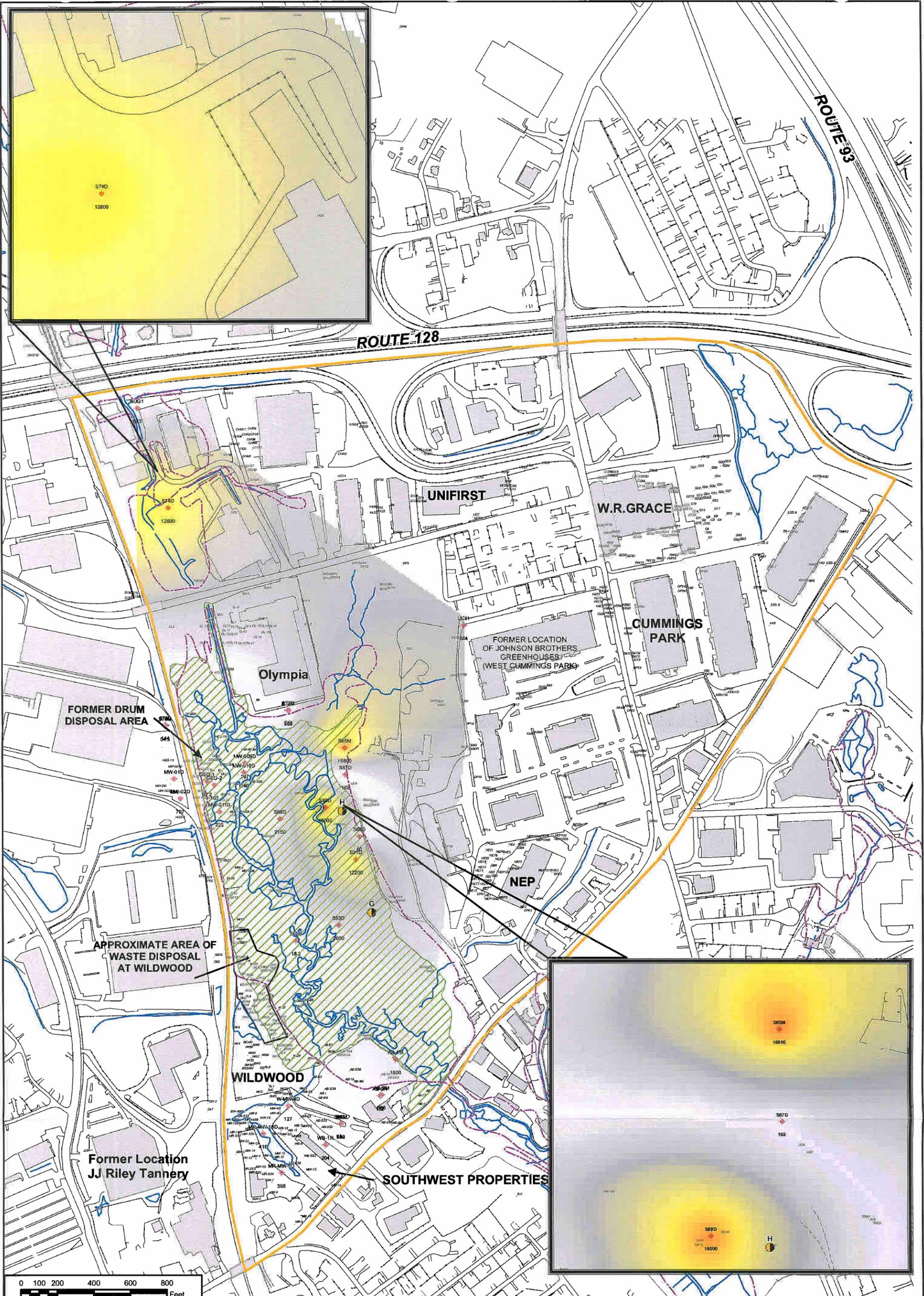


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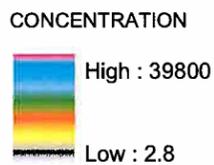
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1985 - 2002



Figure 16C
>50 Foot Manganese
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA



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TRC PROJ. NO.: 02136-0570-0146H

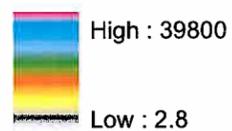
EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values

CONCENTRATION



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
2001 - 2002



Figure 16D
0 - 20 Foot Manganese
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

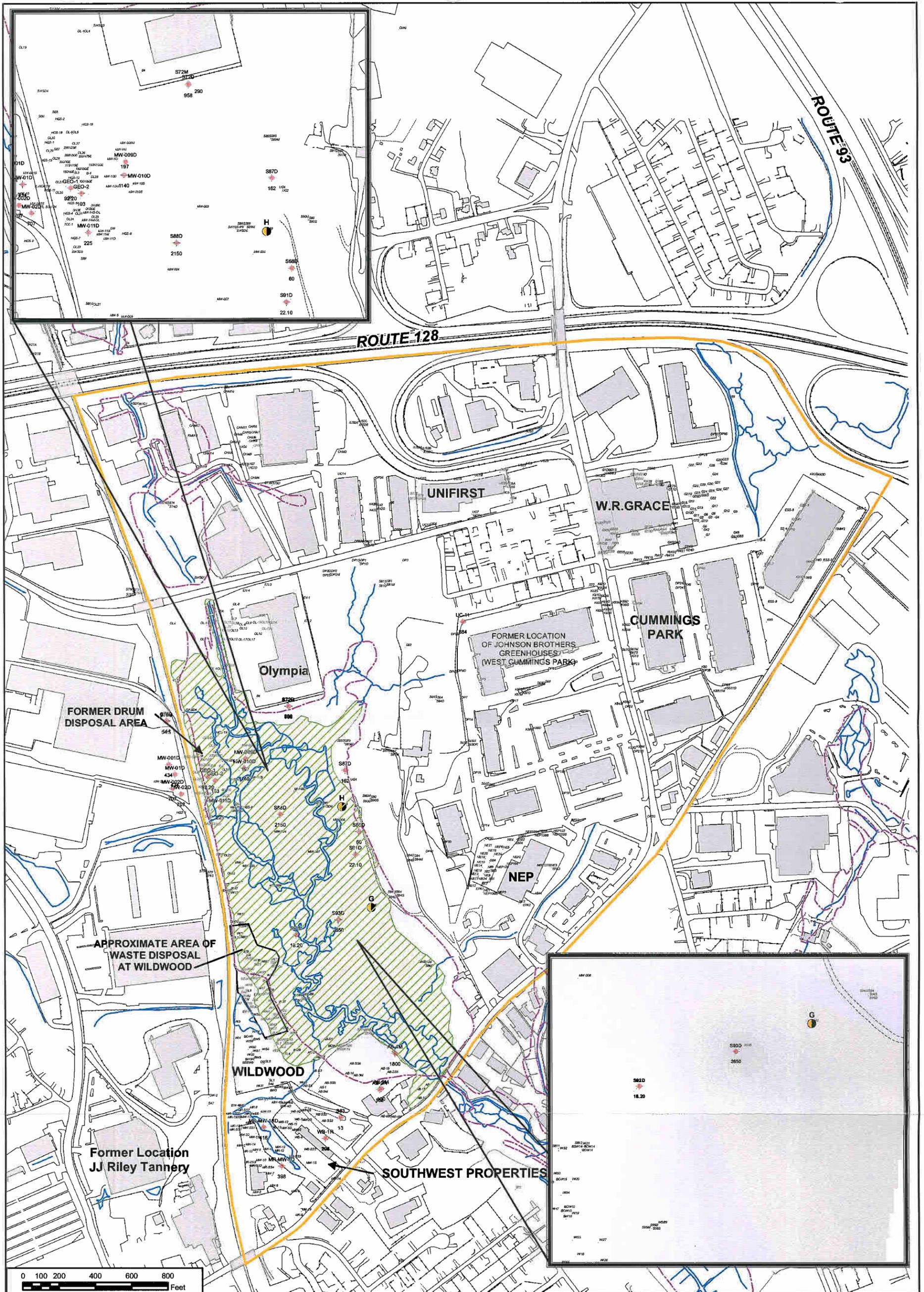
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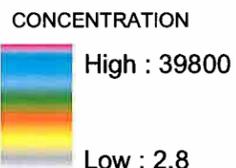
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



Date Range
2001 - 2002



- INORGANIC SAMPLE SITES
- SAMPLE SITES
- WELLS G&H SITE BOUNDARY
- WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Figure 16F
>50 Foot Manganese
Low Flow Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

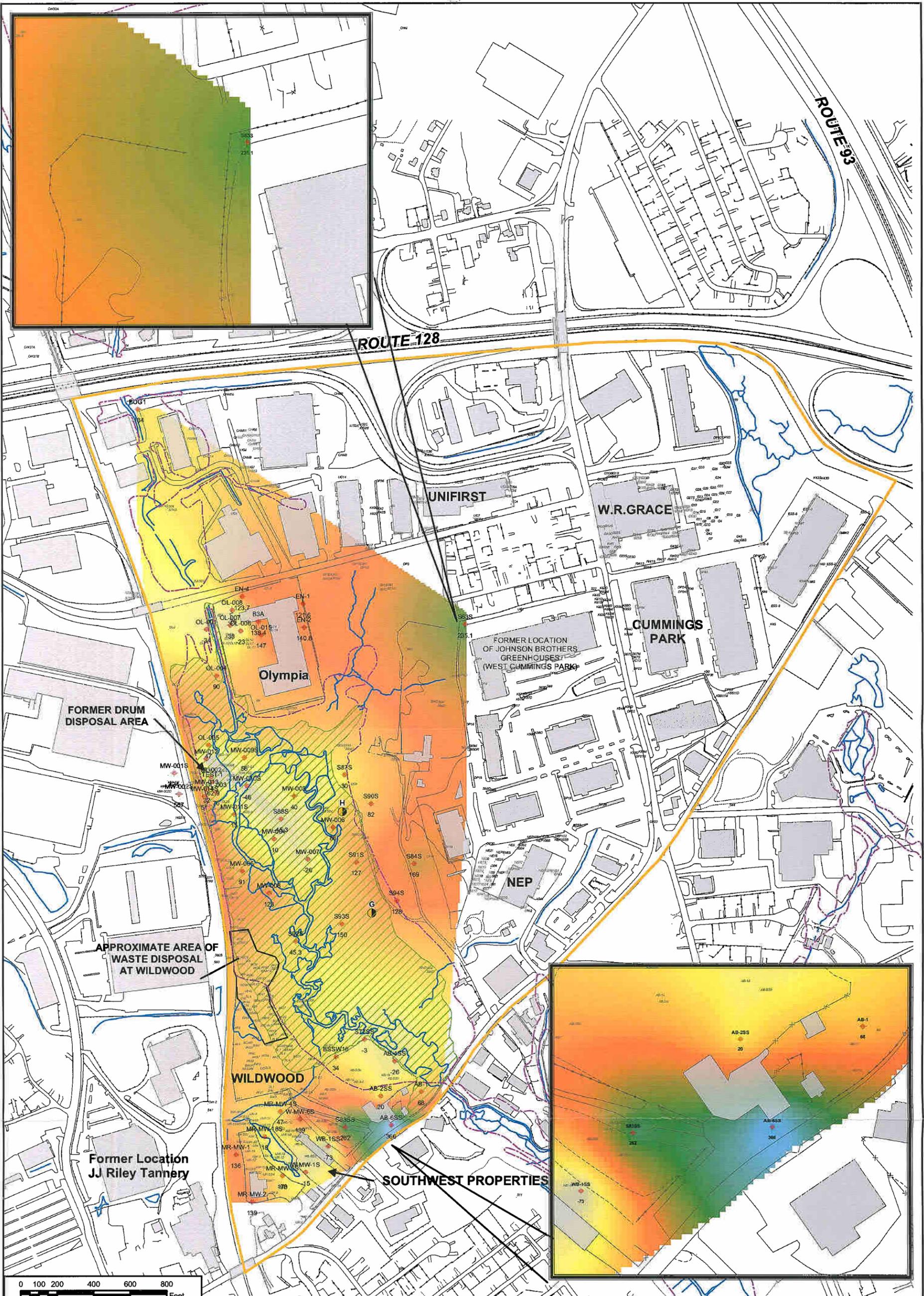
M&E Metcalf & Eddy

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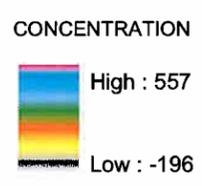
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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- ⊙ SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1991 - 2002



Figure 17A
0 - 20 Foot Oxidation Reduction Potential
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

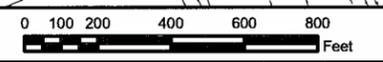
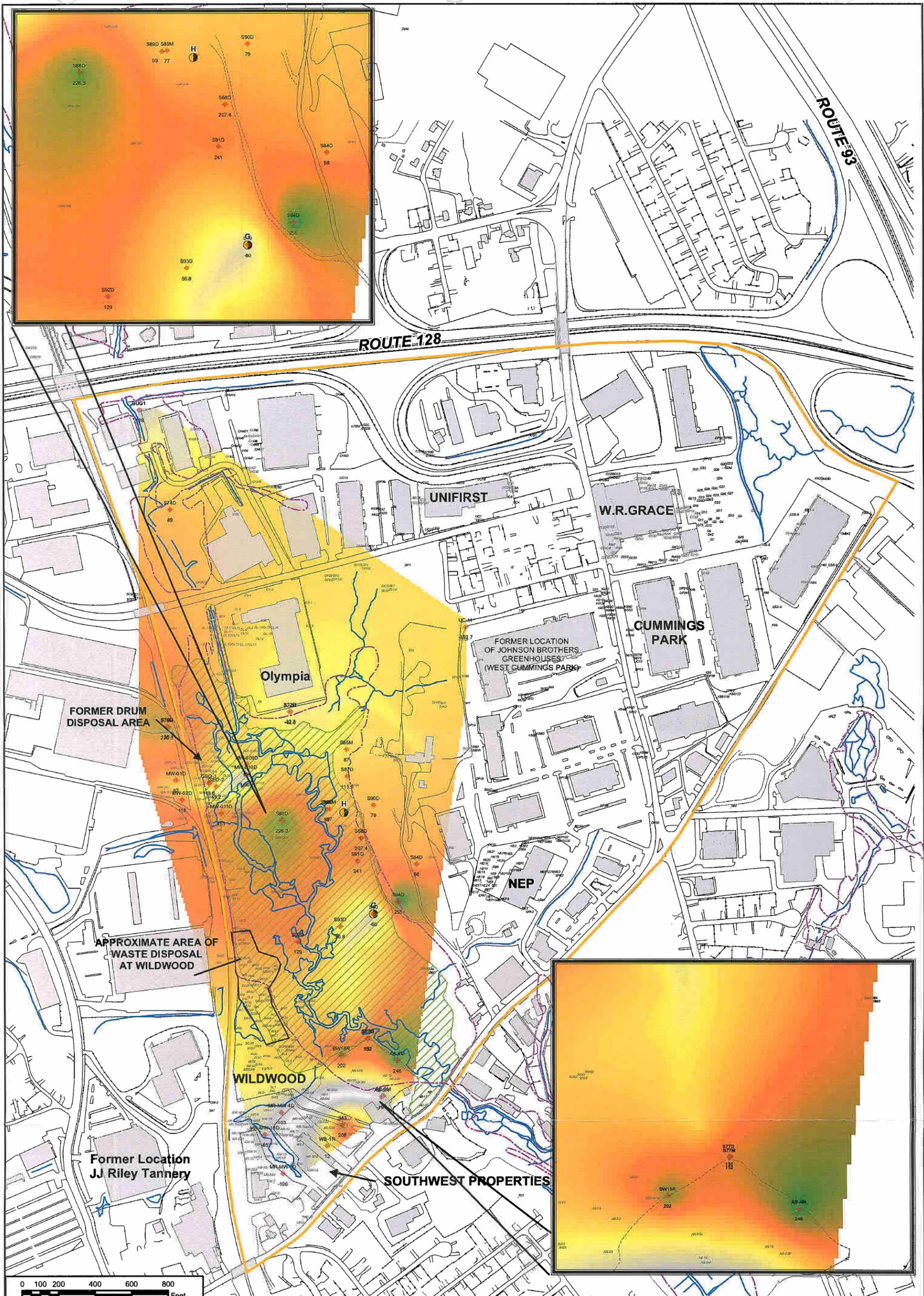


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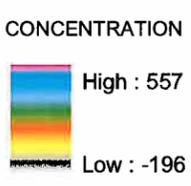
TRC PROJ. NO.: 02136-0570-01461

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOOD PLAIN

Date Range
1991 - 2002



Figure 17C
>50 Foot Oxidation Reduction Potential
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

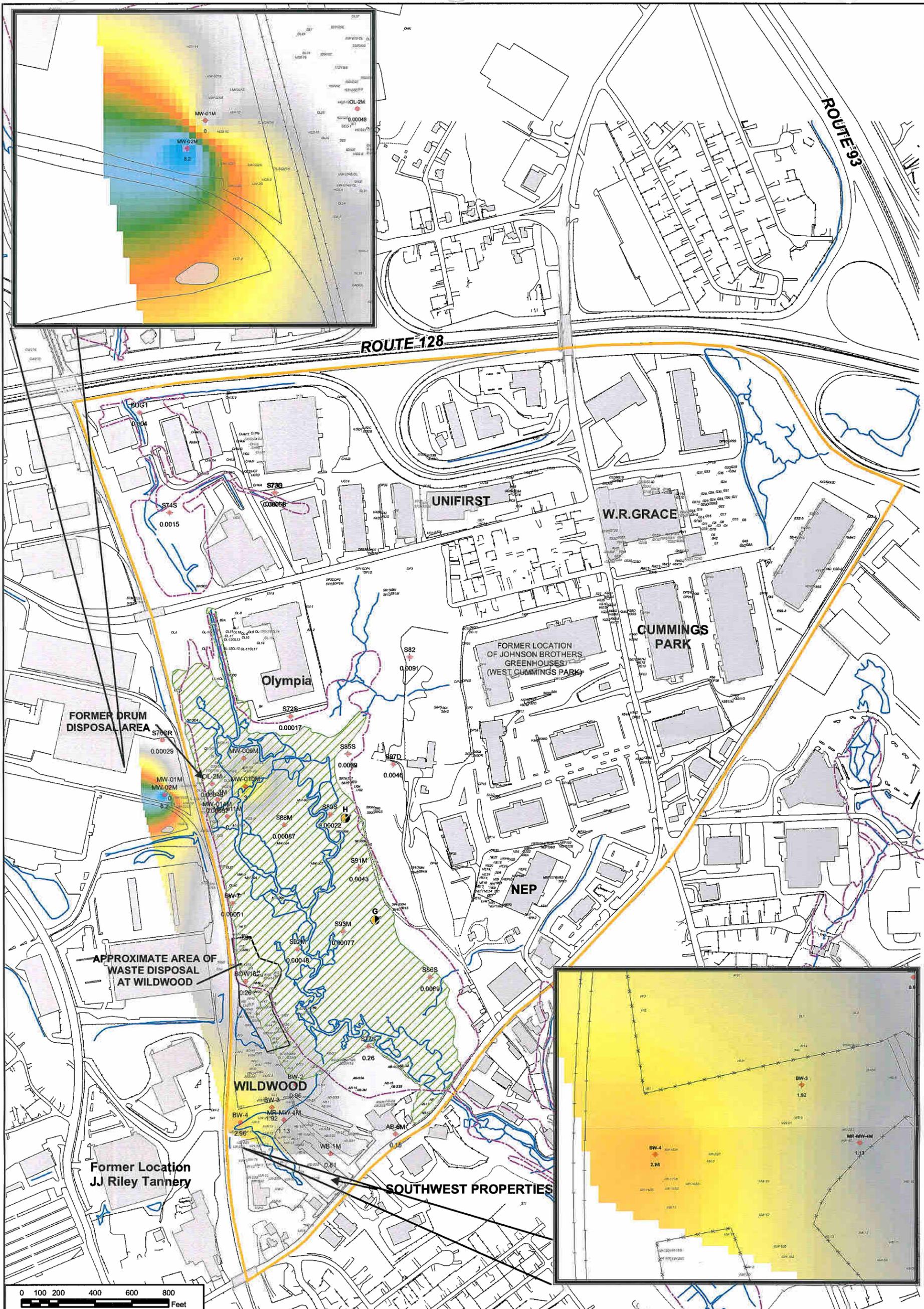


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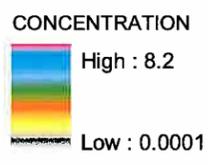
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1993 - 2002



Figure 18B
>20 - 50 Foot Dissolved Oxygen
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

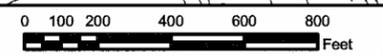
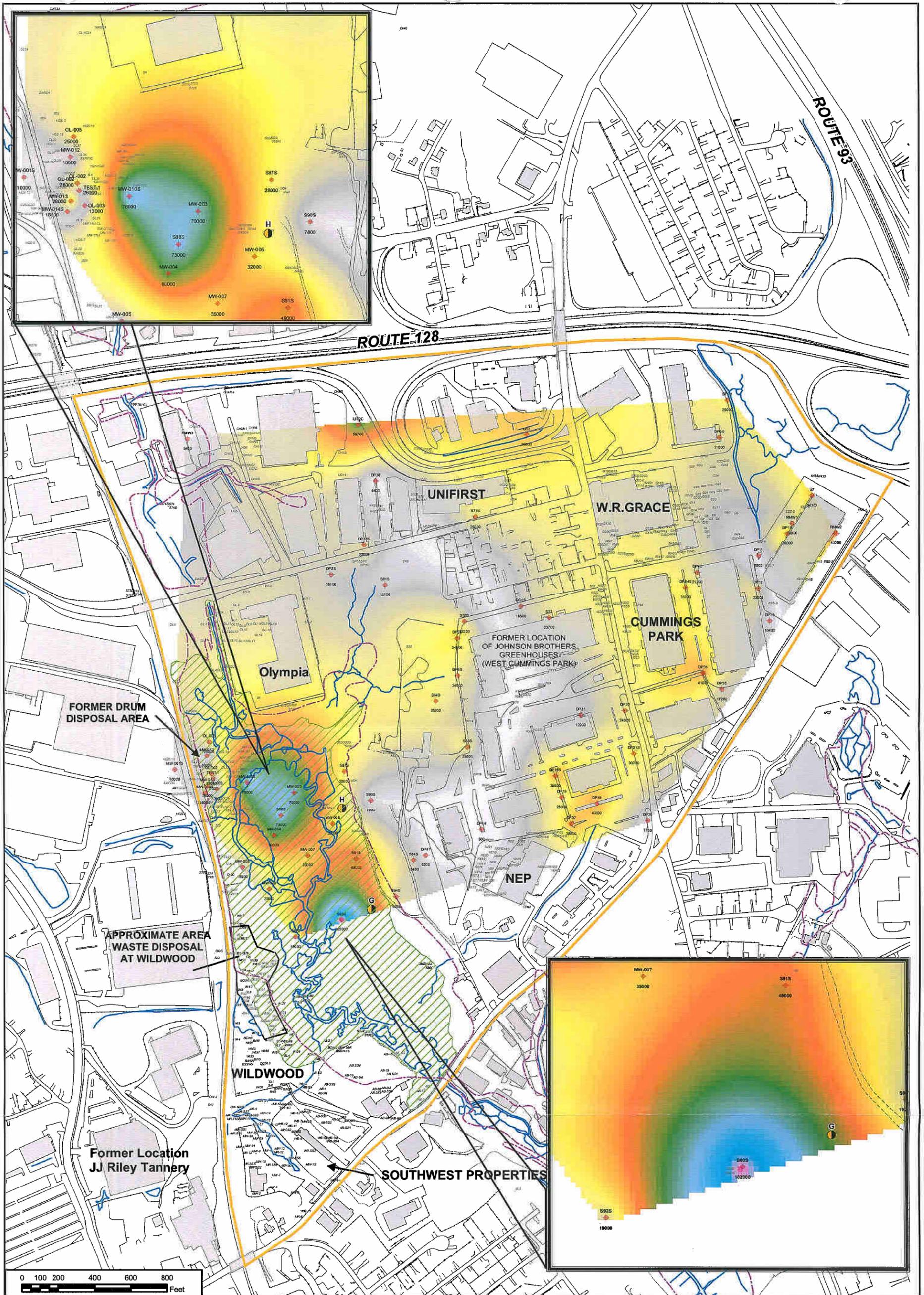


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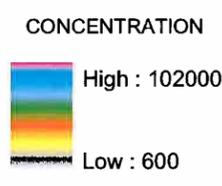
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1991 - 2002



Figure 19A
0 - 20 SULFATE
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

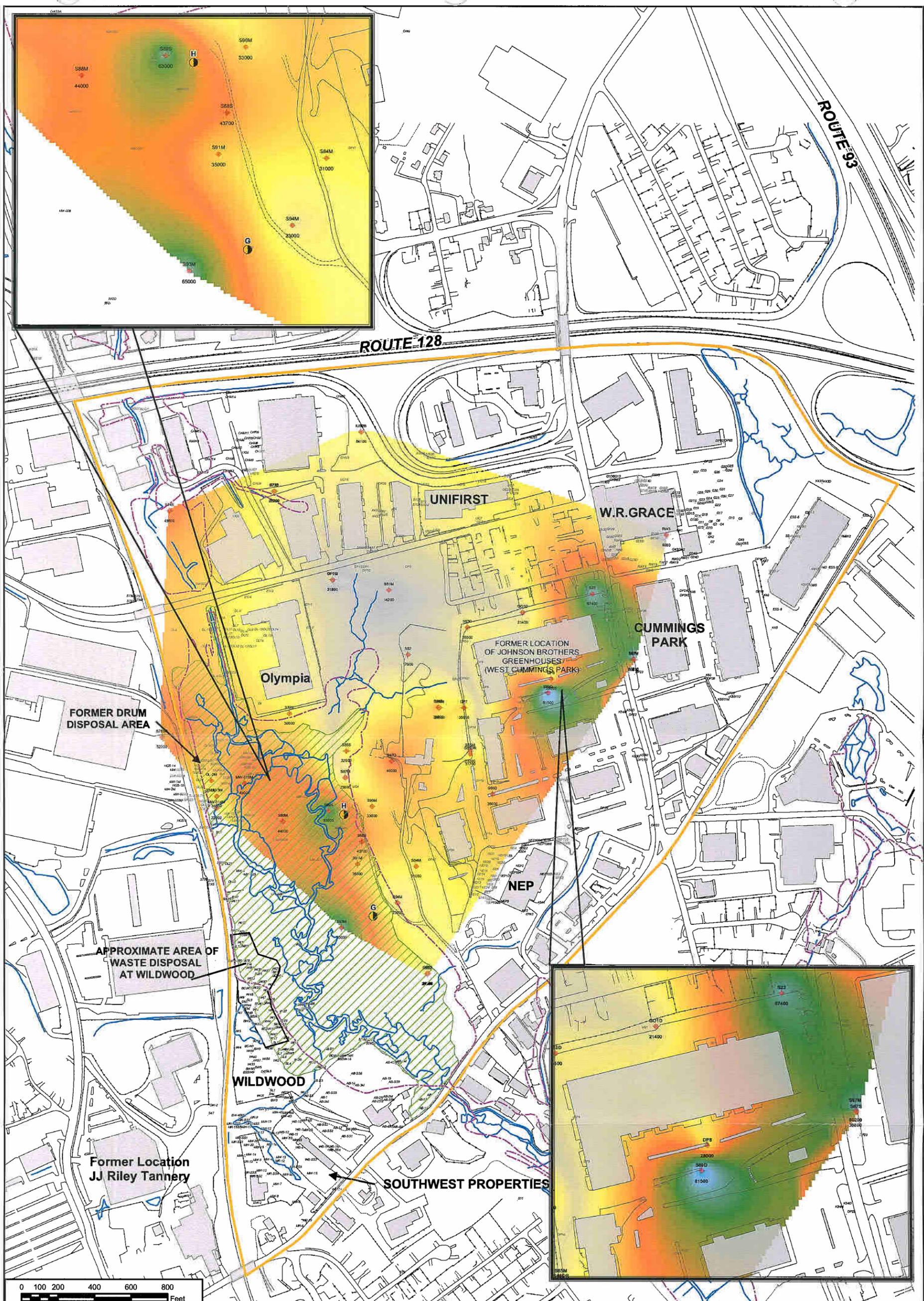


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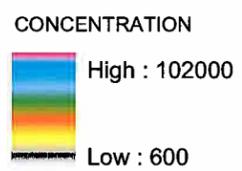
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EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted data are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1991 - 2002



Figure 19B
20 - 50 SULFATE
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA

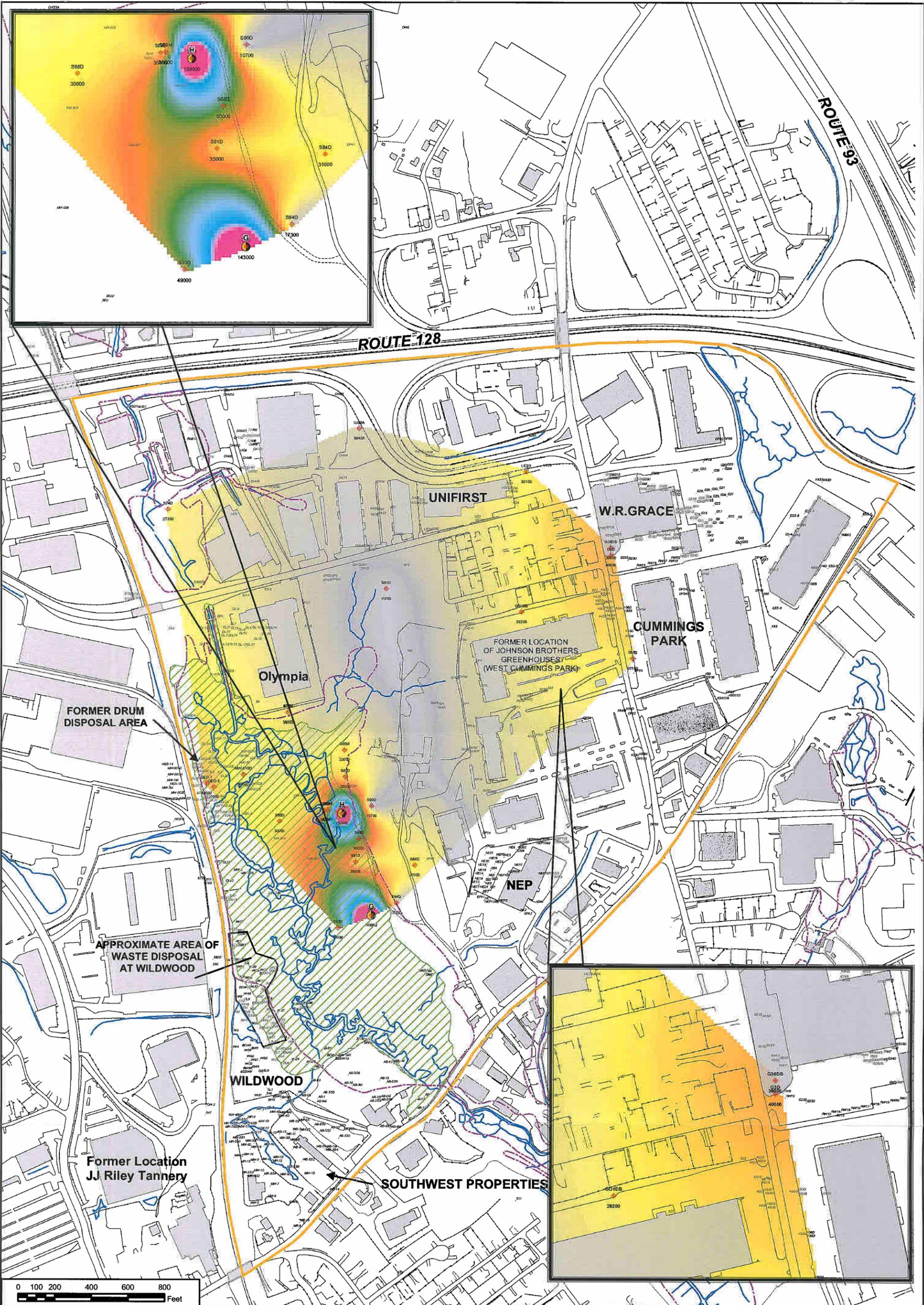


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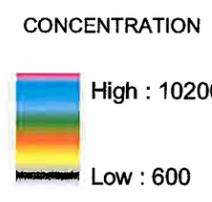
TRC PROJ. NO.: 02136-0570-0146H

EPA CONTRACT NO.: 68-W6-0042

RAC SUBCONTRACT NO.: 107061



All plotted are maximum values



- ◆ INORGANIC SAMPLE SITES
- SAMPLE SITES
- ▭ WELLS G&H SITE BOUNDARY
- ▨ WETLAND EXTENT
- WELLS G&H
- 100 YEAR FLOODPLAIN

Date Range
1991 - 2002



Figure 19C
>50 SULFATE
Groundwater Data
Wells G&H Superfund Site
OU-3
Woburn, MA



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WEST

EAST

Potential short circuiting under pumping conditions

Leaky streambed where peat deposits are not present

Induced flow under pumping brings relatively oxidized meteoric water into aquifer

Aerobic Surface Water As (V) adsorbed to particulates (iron)

Gravel backfill to within 10 feet of surface

Dufrense-Henry coliform remediation excavation and gravel backfill

Land Surface

Water Table

Upper Stratified Drift

Aberjona River

Small Scale Circulation Zone

AEROBIC ZONE (ZONE A)

Upper 1-foot of peat sediments and surface water

- Aerobic conditions
- High microbial activities
- Equal predominance As (V) and As (III)
- High TOC, arsenic, iron and sulfur content
- Low hydraulic conductivity

REDUCTION ZONE (ZONE B)

1 foot to 5 feet bgs

- Highly reducing conditions
- High TOC (% levels)
- Highest microbial activities
- As (III) predominance
- High arsenic, iron, and sulfur content
- Low hydraulic conductivity

SMALL SCALE CIRCULATION ZONE (ZONE C)

- High Iron concentration
- Relatively high As concentration
- Elevated TOC (2000 to 10000 mg/kg)
- Periodic influx of TOC and dissolved As from peat
- Elevated microbial activity (> deeper aquifer)
- Low sulfur content
- Eh-pH regime allows FeOOH solids (-100 mV to +400 mV)
- High hydraulic conductivity

MODERATELY REDUCING/LOW TOC ZONE (ZONE D)

- High Fe sand/gravel
- Moderate reducing conditions (-95mV to -150mV)
- Low concentration As
- Low TOC (130 to 430 mg/kg)
- Low microbial activity
- Eh-pH regime allows FeOOH solids
- Low sulfur content
- High hydraulic conductivity

ZONE "A"

ZONE "B"

ZONE "C"

ZONE "D"

Middle Stratified Drift

Lower Stratified Drift

Fine Sand, Silt, Till, and Bedrock

LEGEND

- Direction of ground water flow
- Small scale circulation zone
- P1** Peat 1
- SP** Sand layers within peat and diatomaceous silt
- P2** Peat 2
- D** Diatomaceous silt

- Low concentration or non detect arsenic in recharge from outer valley
- Relatively oxidized inflow

Cross-Section Sources:
 - USGS, 1989
 - Zeeb, 1996
 - Zone designations and annotations by TRC in green and black

FIGURE 20
CONCEPTUAL MODEL CROSS-SECTION AND FLOW SCHEMATIC UNDER PUMPING CONDITIONS
 WELLS G&H
 OU-3
 WOBURN, MASSACHUSETTS

TRC

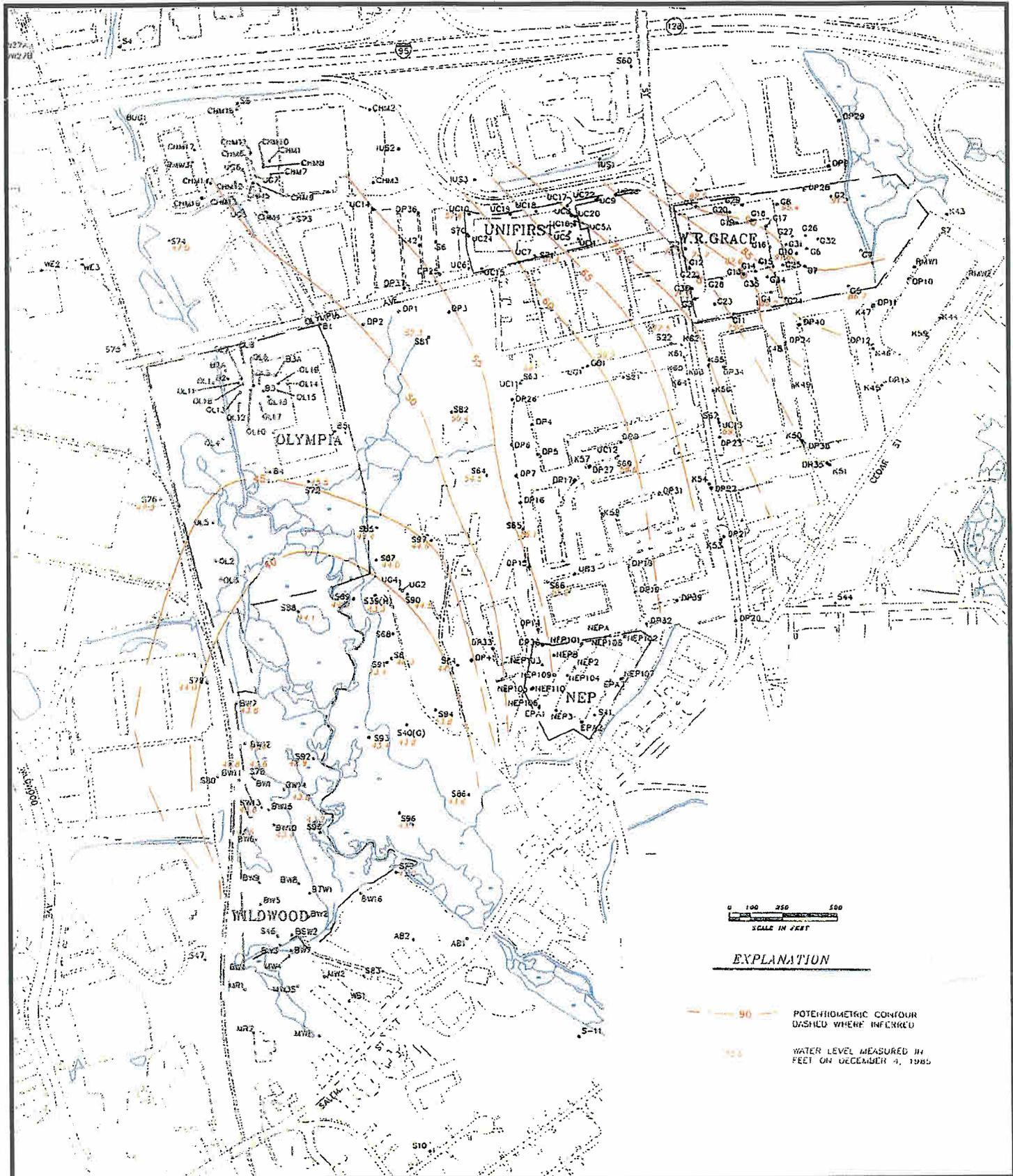
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02136WELLS G&HOU-3CROSS SECT



**FIGURE 21
ESTIMATED WATER TABLE MAP
DERIVED FROM GEOTRANS 1994**

WELLS G&H
SUPERFUND SITE
WOBURN, MASSACHUSETTS



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