

ALTERNATE SOURCE DEMONSTRATION

PREPARED FOR:

**CITY OF SAVANNAH
DEAN FOREST ROAD
MUNICIPAL SOLID WASTE LANDFILL
PERMIT NUMBER: 025-051D(SL)**

PREPARED BY:

ADVANCED ENVIRONMENTAL MANAGEMENT, INC.
2722 SIMPSON CIRCLE
NORCROSS, GEORGIA 30071
PHONE: (770) 242-8282
FAX: (770) 242-8388

Darrell L. Webb, PE, REM
Principal Engineer

William M. Porter
Consulting Hydrogeologist

Patrick M. Gagen, P.G., REM
Sr. Environmental Geologist

March 27, 2007

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1.0 EXECUTIVE SUMMARY

The City of Savannah's Dean Forest Road Municipal Solid Waste Landfill (MSWL) is an active municipal solid waste disposal facility [Permit No. 025-051D(SL)] located on Dean Forest Road in Savannah, Georgia. Groundwater assessment activities have been performed at this facility in response to detected low-level concentrations of inorganic constituents (metals) in groundwater samples from monitoring wells at the facility. Please refer to [Figure 1](#) in [Appendix 2](#) for a facility plan. Early assessment activities indicated that at least a portion of the noted low-level inorganic constituents were being released from the soil matrix (as opposed to being released from the waste unit) due to decreases in groundwater pH levels. However, additional assessment was required to determine the source of the noted pH depressions and to determine if additional constituents were being released from the waste unit or if the waste unit was causing the noted pH depressions.

Based on groundwater flow away from the waste units and an assessment of surrounding land use, no obvious off-site sources were identified. Therefore, the waste unit was initially assumed to be the source of the noted impacts (pH depression and low-level inorganic constituents). As a result, a thorough assessment was conducted to determine if and how the waste mass could depress pH levels and/or release inorganic constituents to the groundwater beneath the facility. The potential impact mechanisms identified included leachate- or landfill gas-driven releases from the waste units.

After identifying the potential impact mechanisms, AEM evaluated site-specific data to determine how the identified mechanisms could explain the impacts noted in the monitoring system. Examples of site specific data include leachate testing results, landfill gas monitoring results, subgrade testing following waste excavation, groundwater flow rates and direction, and the groundwater monitoring data itself. Based on this evaluation, AEM determined that none of the potential mechanisms could describe the impacts noted at the site. Therefore, AEM concluded that the noted impacts were not migrating from the waste unit at this facility. With this conclusion, the scope of this assessment project shifted back to an alternative source assessment project.

AEM performed extensive research in an effort to identify a potential alternative source that could account for the impacts noted at the facility. The presence of severe pH depressions at the facility led AEM to suspect that naturally-occurring conditions known as Acid Sulfate Soils (AS soils) could account for the noted impacts. AS Soils contain sulfate in a reduced state that, upon exposure to air, oxidizes and produces Sulfuric Acid. This acid then lowers the pH of the groundwater significantly and leaches metals from surrounding soils.

The presence of AS soils has been confirmed at the facility through field and laboratory testing. In AEM's professional opinion, oxidation of Acid Sulfate Soils and the associated depression of groundwater pH are the source of the impacts noted

at this facility. As AS Soils occur naturally, these conditions represent an alternate source.

2.0 BACKGROUND

The Dean Forest Road Landfill was permitted by the Environmental Protection Division in 1982, and disposal operations began in March 1984. The facility initially accepted putrescible waste along with construction and demolition debris from within the City of Savannah. The waste stream presently includes municipal waste combustor ash from the Savannah Resource Recovery Facility (SRRF), privately owned and operated on behalf of the City's Resource Recovery Development Authority. Additional facility history is provided in [Appendix 4](#).

Groundwater monitoring was initially conducted in six wells located on the southeast corner of the site. That system was replaced in 1993 by a network of 23 downgradient wells and two background wells. Additional wells have been installed since 1993 with a total of 33 wells being monitored as of October 2005. Additionally, five Surface Water sampling stations have been monitored semi-annually since 1993 and Methane monitoring has been conducted quarterly. Groundwater detection monitoring data from 1993 through October 2005 are tabulated in [Appendix 10](#) for reference. No data from the pre-1993 six-well system were used in preparation of this report. The groundwater and surface water sampling locations have also been used, in conjunction with a network of piezometers and stream gauges, to obtain data on groundwater flow directions and gradients at the facility.

Beginning in 1996, elevated metal concentrations detected in well GWC-11, coupled with a lack of obvious off-site sources, led the facility into assessment monitoring. Although metal concentrations in this well eventually declined, metal detections in other wells at the site increased and required additional assessment. Initial assessment activities were designed to identify the nature of the impacts noted. As inorganic constituents naturally occur in site soils, the early assessments attempted to determine if the noted detections were due to migration of constituents from the waste unit or release from the soil matrix. Early observations also noted that pH levels in impacted wells were significantly depressed. Although the depressed pH was an obvious potential source of the elevated metal detections, the landfill could not be ruled out as the source of the pH depressions and, therefore, assessment activities continued with the impacts defined as the elevated metals and the pH depressions.

More recent assessment activities focused on analyzing facility history and gathering additional subsurface and analytical information in an attempt to identify any processes that might cause the noted pH depressions at the facility or migration of the noted metals from the waste units. These recent assessments led AEM to conclude that the noted pH depressions are naturally occurring and the source of the noted inorganic impacts. AEM arrived at this conclusion after thoroughly analyzing potential ways that the waste units could be causing the pH depressions or the inorganic impacts noted. In our analysis, none of the potential ways the waste units could impact groundwater was supported by the data available from the facility.

Based on the conclusion that the landfill was not the source of the noted impacts, AEM performed additional assessment to identify other possible sources. These additional assessment activities led to discovering Acid Sulfate Soils (ASS) at the facility. AEM concluded that these soils are the source of the noted impacts and, therefore, AEM has prepared this Alternative Source Demonstration (ASD) to document the assessment activities conducted and the results that led AEM to this conclusion.

The ASD begins by defining the impacts noted at the facility. Subsequent sections address the potential for off-site sources of the noted impacts and present data from the site used to assess the potential for the waste unit to cause the noted impacts. Alternative sources are then discussed and conclusions presented.

This ASD complies with the Georgia Department of Natural Resources Environmental Protection Division regulations regarding solid waste management, specifically section 391-3-4-.14(30)(b) of the Georgia Code.

3.0 GROUNDWATER IMPACTS

Groundwater impacts at this facility have been identified through previous monitoring and assessment activities. Following early redevelopment activities to minimize the effects of turbidity on sample quality, assessment activities were initiated in response to elevated concentrations of Beryllium, Cadmium, and Selenium in wells GWC-10, GWC-11, GWC-12, GWC-17, and GWC-17A. As documented in the July 7, 1998 Assessment Monitoring Report, only the Cadmium concentration in GWC-11 was deemed a potential groundwater impact at that time. However, by 2002, additional concerns at the facility led to completion of an Assessment of Corrective Measures Workplan. The workplan included a detailed analysis of historical detection data for the facility and concluded that additional assessment activities were required for pH, Barium, Beryllium, Cobalt, Lead, and Nickel in wells GWC-2, GWC-3, GWC-5, GWC-10, GWC-11, and AMW-1. These parameters and locations define the groundwater impacts for this facility.

A history of assessment activities conducted at this facility is included in [Appendix 5](#) for reference.

4.0 SOURCE ANALYSIS

Municipal Solid Waste Landfills are potential sources of impacts to groundwater. It is for this reason that routine monitoring is required at such facilities. However, there is always the potential that impacts detected as a result of this monitoring are actually caused by some other source. Therefore, a thorough assessment typically includes identifying potential offsite sources for noted impacts. This is accomplished by identifying surrounding property uses and assessing the potential for those uses to cause the noted impacts.

4.1 Potential Offsite Sources

At this facility, surrounding property uses are limited primarily to residential uses. The exception is a heavy equipment rental facility located adjacent to the northeast portion of the facility. Offsite sources were dismissed early in this assessment due to the following factors:

- The only readily identifiable potential offsite source of inorganic impacts (the rental equipment facility) was located at a significant distance from the initial area of concern (GWC-11) and did not appear to impact more proximate monitoring locations.
- The groundwater flow is away from the waste unit and toward the impacted monitoring locations. As such, impacts from offsite sources are unlikely.

4.2 The Landfill as a Potential Source

After eliminating potential offsite sources of the noted impacts, AEM turned its attention to the waste unit. In general, municipal solid waste landfilling activities can impact groundwater in one or both of two ways – leachate or gas impacts. Leachate is produced when liquids (either contained in the waste or from rainfall) percolate through the waste mass and carry contaminants down to the water table beneath the waste unit. Gas is produced in the waste unit as putrescible waste decomposes. This gas primarily includes Methane, Carbon Dioxide, and trace amounts of various contaminants. As the gas is produced during the decomposition process, a pressure gradient is created within the waste unit that drives the gas away from the waste unit boundaries and into the surrounding formation. While leachate directly impacts groundwater, gas impacts may be direct, indirect, or both. Direct gas impacts occur when contaminants migrating in a gaseous phase are absorbed by groundwater in contact with the gas. An indirect gas impact occurs when Carbon Dioxide reacts with groundwater to create a weak, carbonic acid that may leach minerals from the surrounding formation.

AEM conducted an extensive study to evaluate the potential impact mechanisms associated with landfilling activities. This study included analyses of waste characteristics, subwaste soil testing data, groundwater and surface water monitoring data, leachate monitoring data, site geology and hydrogeology, disposal history, and other available data. Results of this study are detailed in the following two sections.

4.2.1 Assessment of Potential for Gas Impacts

As described above, gas is generated in a waste mass as a result of decaying putrescible waste. At this facility, however, a significant fraction of the landfilled waste is non-putrescible. For example, 2005 disposal records indicate that approximately 35% (by weight) of the landfilled waste consisted of ash. As such, gas generation is expected to be reduced at this facility. Low gas generation rates are also indicated by the following data:

- Methane monitoring is conducted at this facility quarterly. To date, no elevated methane concentrations have been detected.
- All groundwater wells were sampled for Methane, Carbon Dioxide, and Oxygen on July 27, 2006. Methane and Carbon Dioxide concentrations were all below 0.1% - within the error range for the instrument. (refer to [Table 12](#) in [Appendix 1](#)).
- The lack of VOC impacts in leachate and groundwater analyses at the facility suggests low gas generation.

Based on these site-specific data, there is no indication that the waste unit at this facility produces sufficient gas to cause the impacts noted.

4.2.2 Assessment of Potential for Leachate Impacts

The potential for leachate generated in the waste unit to cause the noted impacts was assessed through a thorough evaluation of the following site-specific data:

- Two general categories of waste have been placed in the Dean Forest Road facility: municipal solid waste (MSW) and ash from waste combustion. While MSW is typically not tested prior to disposal, leaching tests are routinely conducted on representative ash samples prior to landfilling the material. Results from these analyses were used to evaluate the potential for the waste to release the constituents of concern.

- Site soils beneath trenches were tested for metals after the MSW and ash were removed for placement in lined cells. These data were used to assess the likelihood of metals having been leached from the waste and infiltrating to the water table.
- Analysis of composite leachate samples from Lift Station 150 took place monthly or more frequently from 1995 to 1998, and semi-annually thereafter. Leachate analyses included inorganics and pH. Surface water stations SWC-1 through SWC-5 in the perimeter ditch system were sampled semi-annually beginning in October 1996. Surface water analyses also included inorganics and pH. These data were evaluated to assess actual leachate characteristics.

4.2.2.1 Waste Characteristics

Ash from the Savannah Resource Recovery Facility has been tested by the EP Toxicity Test (prior to circa 1988) and later the Toxicity Characteristic Leaching Procedure (TCLP). The results of earlier annual testing were not found, and the available data span the period from July 1995 to the present. By 1995, the ash was being treated with a proprietary process (WES-PHix[®]) to minimize leaching, and thus the results at hand would not accurately represent the characteristics of earlier ash.

TCLP results for conditioned ash are summarized in [Table 9](#). Fly ash and bottom ash are mixed prior to disposal, but the percentages of each are not known. The mean concentrations in both fly ash and bottom ash are generally low and no metal was consistently detected. While characteristics of older (pre-treatment) ash cannot be inferred from these test results, the frequencies of detection may provide some insight into which metals were most abundant and persistent prior to treatment and thus more likely to have been leachate constituents.

Barium has been detected more often than any other metal in the treated ash (69% of the time in fly ash and 14% in bottom ash). The only other metals that have been frequently detected are cadmium (15% in fly ash, 6% in bottom ash), lead (10% in fly ash, 9% in bottom ash), and chromium (25% in fly ash, although the average concentration has usually been at or near the detection limit).

The Facility History ([Appendix 4](#)) references the following TCLP results for the older, unconditioned ash:

- May 1988, exceeded TCLP limits for lead (>5.0 mg/L) and cadmium (>1.0 mg/L).
- July 1989, TCLP levels for barium in individual samples up to 204 mg/L, lead up to 54 mg/L, cadmium up to 4.7 mg/L.
- May 1993, fly ash tested separately had cadmium concentrations in individual samples up to 6.3 mg/L.

The ash placed in trenches prior to construction of the lined facility might or might not have generated leachate with any significant concentrations of metals. However, the above data suggest that barium, cadmium and lead would be the most likely constituents.

With regard to testing results, it is believed that the earliest results pertained to grab samples rather than composite samples, and thus may not accurately represent the overall characteristics of ash at that time.

It is also important to recall that the TCLP procedure involves leaching by a buffered acetic acid solution maintained at a constant pH of 5.0, emulating the worst-case situation of co-disposal in a landfill receiving organic waste under anoxic conditions. The older EP Toxicity procedure was even more aggressive with regard to metals extraction. These tests do not provide a good indication of actual leaching potential under many disposal situations.

Numerous studies have demonstrated that the concentrations of metals leached from ash are very strongly dependent on pH, being orders of magnitude less in many cases for an increase of one pH unit within the pH 4 to 7 range. The pH of pore water also depends on the buffering characteristics of the waste. Solid waste combustion ash generally has a higher buffering capacity than coal ash (He et al, 2003), tending to drive the pH higher. The segregation of ash in separate trenches at this site would have resulted in higher pH of pore water and far less leaching than suggested by the TCLP and EP Toxicity results.

4.2.2.2 *Subwaste Soil Characteristics*

Subgrade testing provides another source of information about the potential for metals leaching from landfilled waste. The subgrade testing was done after waste was removed from the Southwest quadrant and the four westernmost trenches in the Southeast quadrant. The testing included soil beneath 6 trenches that had been filled with ash and 11 trenches (plus one wet-weather trench) that received MSW.

Metals may be retained on soil to differing degrees depending on the metal, the types of clay present, and environmental conditions. Nonetheless, it is unlikely that significant amounts of leachate could have infiltrated through the soil column to underlying groundwater without leaving behind some evidence of their passing.

Table 10 presents the results of subgrade testing. It is emphasized that these results are concentrations of metals in soils expressed as mg/kg, and not the results of leaching tests such as TCLP. Also, all the metals in this table occur naturally in soils and sediments. Included on this table is the range of reporting limits and average reporting limit for the metals where these were not detected. This can be compared against the average levels detected for each metal and the typical values for soils to determine whether analytical limits could be masking some results.

Table 11 is a summary of the subgrade testing results. Omitted are metals that had no detections or very few, excepting cadmium (one detection only, in ash) because this metal is of interest in monitoring results. The percent of detections are shown in this table, along with average values for the ash trenches, the MSW trenches, and for all trenches together. Non-detects were not considered in computing the averages. Two additional rows in this table present the average metals concentrations for shallow water argillaceous (clayey) sediment and sandstone (taken from Salomons and Förstner, 1984, Table 42). These two sets of averages, based on very extensive data from research literature, provide a range that could typify either clayey or sandy soils at the site.

Comparing the detection limits (reporting limits) for instances where the metals were not detected against the subgrade averages and the average values for sediments in general, there appear to be three categories:

1. For arsenic, beryllium and possibly cadmium (only one detection), some concentrations higher than the sand average but less than the clayey sediment average might have been hidden by the detection limit.
2. For cobalt, copper, lead and nickel, some concentrations near the subgrade averages might have been hidden by the detection limit, but those levels would have been below the values for average sandy or clayey sediments. Therefore comparisons would not be significantly affected.
3. For barium, chromium, vanadium and zinc, detection limits were substantially lower than subgrade averages and averages for typical sediments, and thus could not affect comparisons.

The averages for subgrade metals appear to fall within or below the expected values for natural sediments in nearly every case. The subgrade concentrations appear to be closer to those expected for uncontaminated sand than for unaffected clay. Site cross-sections indicate that material at the surface in this area is mostly clayey sand, possibly with some areas of silty sand, so trace element concentrations at the lower end of the range make sense. Clayey sand, however, would contain enough clay minerals to retain additional metal cations if they reached the soil in high concentrations from leachate.

Even though the arsenic average for ash trenches is disproportionately influenced by one value, it could indicate occasionally higher levels of this metal that might locally leach into the soil. This is one case where levels are slightly higher than expected in the absence of disposal, although this is certainly within the range of natural variation.

If the average value for sand is used for comparison, the subgrade cadmium appears to be one order of magnitude higher, but that conclusion would be very misleading. Subgrade cadmium in this table is represented by one sample only, and cadmium was not detected in any of the other samples. In reality the results suggest that very little cadmium ever passed through the soil column at this site.

The beryllium average in the subgrade is disproportionately affected by one value out of four from ash trenches – the other three are similar to those in the MSW trenches. Beryllium was detected in only 11% of the ash trench subgrade samples, compared to 27% of the MSW trench samples. Thus there is no indication here of significant beryllium releases from either ash or MSW.

Cobalt in the subgrade is higher than expected for sand, lower than expected for clayey sediments. Since most surface material here is clayey sand, the results fall within the range of expectations for unaffected soils and this data does not seem to present evidence of releases.

Of the remaining metals, barium, chromium and copper subgrade averages are considerably lower than the averages for average sandy sediment. Lead, nickel, vanadium and zinc are all below but reasonably close to the expected values for sand.

We may be able to infer something more about leaching from ash and MSW by comparing the average subgrade concentrations in [Table 11](#). Considering only the subgrade testing averages where a specific metal was detected at least 50% of the time, barium, copper and zinc might possibly be slightly higher at ash trench sites, but chromium and vanadium are higher at the MSW trench sites, and lead is essentially the same. Thus there is no clear difference. Averages for metals that were detected in less than 50% of the samples are generally less accurate, and in any case suggest that large concentrations of those elements never passed through the soil in leachate. If barium, cadmium and lead were being leached out of the ash at higher rates than for municipal waste, some traces of them over background levels would be expected.

In summary, TCLP testing implies a possibility that some of the older, untreated ash might have a potential to release barium, cadmium and lead, but the subgrade testing suggests that this did not actually occur. Concentrations in the soil are similar to or lower than expected for unaffected natural clayey sand sediment. There are no elevated levels of metals that might be associated with the ash, nor any consistent indication of higher levels of those metals in soils underlying the ash trenches when compared to the MSW trenches.

4.2.2.3 *Leachate/Surface Water Characteristics*

Data from ditch monitoring may provide some indirect information about leachate properties and the source of metals. Water in the ditches consists of a mixture of runoff plus groundwater discharge. The groundwater discharge could include water that has infiltrated through the waste, and thus both runoff and seepage to the ditch may have properties of leachate.

[Figure 28](#) shows pH data for two wells of concern on the west side of the landfill (GWC-11 and AMW-1) along with pH data for the leachate composite and the nearest (downstream) surface water monitoring station, SWC-5. [Figure 29](#) presents similar data for wells on the east and south sides (GWC-2 and GWC-5), the leachate composite, and the nearest surface water stations, SWC-3 and SWC-4 (see [Figure 1](#) for locations).

Several observations can be made from these plots. The typical pH of the leachate composite has been near neutral, mostly occupying the range between 6.0 and 7.5 units. The pH of ditch water has varied a great deal more, but the average remains within the same range as the leachate composite. After 1997, the average groundwater pH has been much lower than either the ditch samples or the leachate. The difference in pH between surface water and groundwater has usually been 1.5 to 2.5 units for comparable dates. Nor is the lower pH confined to wells downgradient from the landfill; background wells GWC-3 and GWC-4 displayed a similar pH decrease in 1996-1997, and the average pH was between 4 and 5 units until recently. Thus, the lower pH of groundwater is not caused by low-pH leachate.

If the leachate is close to neutral and groundwater is more acidic, a process completely unrelated to leachate must be responsible for decreasing the groundwater pH. That process would also be likely to release metals from clays.

The possible effects of precipitation are discussed in [Appendix 9](#), but for the present it may be noted that pH decreases in samples from ditch monitoring appear to correlate to some degree with preceding periods of heavier

monthly rainfall. Decreased ditch pH is likely to represent an influx of groundwater with a lower pH. There are several reasons why the response of groundwater pH at specific wells may be different, and these will be explored in a subsequent section of this report.

In summary, actual leachate and surface water results indicate that pH levels in leachate are not depressed and, therefore, that significant leaching of metals from landfilled waste is unlikely.

4.2.2.4 Analyses of Impacted Locations

The preceding sections provide a substantial amount of data relating to the characteristics of the landfilled waste at this facility and the potential impacts that could be caused by the waste. In this section, these data will be compared to data from the impacted locations to assess the likelihood that the impacts are due to the waste unit.

GWC-11

This well was very close to trenches on the western end of the Southwest quadrant that received segregated disposal of fly ash and bottom ash between 1988 and 1991. Details are provided in [Appendix 4](#) (Facility History) and the map of trench locations in [Figure 2](#). The ash in those trenches was placed prior to implementing the WES-PHix® process, and thus the metals of greatest concern with regard to leaching would be barium, cadmium and lead.

Reviewing the detection data at this location, the metals that were elevated between late 1996 and late 1998 were barium and cadmium (TCLP metals) plus beryllium, cobalt, nickel and selenium. Lead was detected on only one date during that period. If lead were a constituent of leachate, however, it might have been selectively bound to native clay particles and thus fixed or its migration attenuated. However, subgrade testing in this area did not indicate the presence of elevated lead levels.

With the exception of the absence of lead, the correspondence between detected metals and ash constituents is fairly good, and the distance between the well and the ash trenches is small. As seen in [Figure 23](#), the sediments between the ash

and the water table would have been clayey fine sand, and that would have favored infiltration to a greater degree than clay. However, testing of subgrade soils in the vicinity, including three locations in the footprint of the trench closest to this well, provide no indication of metals having leached to the subsurface.

Taking all of this into consideration, a landfill source for metals in this area is considered “possible”. However, a landfill source does not explain why concentrations suddenly increased and abruptly decreased two years later.

As noted before, the elevated concentrations ceased after October 1998, and the waste has been removed from this quadrant. Subgrade testing has been conducted to confirm that no waste residues remain. A liner will be placed over the area of the former ash trenches, further reducing any possibility of infiltration or residual constituents being washed downward. Therefore, this area is no longer of concern with regard to future releases.

AMW-1

This well (along with AMW-2) is positioned on the opposite side of the ditch from the ash-filled trenches mentioned above. The implications of stratigraphy and water levels, seen in [Figure 23](#), in connection with the potential for groundwater flow to trenches in this area is discussed in [Appendix 7](#) (Hydrogeological Assessment).

Except during periods of very high or very low water table, the shallow groundwater will be flowing through sandy clay as it moves toward the ditch, and very low flow rates on the order of 0.1 to 0.2 feet per year ([Figures 11 and 12](#)) would be expected. If leachate from the ash produced a plume with sufficient depth in the water column, some of that plume might flow beneath the ditch through the underlying clayey sand. Using the properties of clayey sand and the gradient to the west, an average flow velocity 0.55 feet per year would pertain to the deeper flow, with a maximum of 2.3 feet per year.

AMW-1 is about 150 feet from the nearest waste disposal, and even if the highest flow rate is assumed, there would not have been nearly enough time for constituents to reach this

well between 1988 (the earliest disposal in trench 28) and 1998 (when the well was installed).

In addition, there is a poor correspondence between TCLP metals of concern and those seen in AMW-1. Cadmium, barium and lead were the likely products of ash leaching; although barium is present at AMW-1, cadmium and lead have not been detected here, nor at AMW-2.

There are also considerable differences between the metals detected at GWC-10 and GWC-11 on the inside of the ditch, and the metals found in AMW-1 and AMW-2 on the outside of the ditch. For example, beryllium, nickel and cobalt have consistently been detected at AMW-1 since it was installed in 1998; these metals have not appeared in GWC 10 (which is opposite from AMW-1) apart from the initial monitoring periods, and were found at GWC-11 only during the two-year period of elevated concentrations..

Based on the anticipated low flow rates, the role of the ditch as a barrier to shallow migration, and dissimilarity of detected metals from those in the ash and inside the ditch, a landfill source for metals in this area is considered “unlikely”.

GWC-2/GWC-3

These wells are situated next to the locations of trenches 44 through 50 in the Southeast quadrant. Those trenches never received fill prior to installation of a liner in 1995 (see [Figure 2](#)). Toward the west, the closest unlined disposal sites were at trenches 43 (MSW) and 42 (ash). Thus the nearest unlined waste disposal would be 650 to 700 feet west of GWC-2 and GWC-3, and would have contained fill for only a few years - the period from 1991 until waste was excavated in late 1996.

The site cross-sections indicate that shallow saturated materials between filled trenches to the west and GWC-2 would be mostly or entirely sandy clays. Assuming that groundwater had ever flowed directly to the west, the expected flow velocity through sandy clays might average 0.12 feet per year, with an absolute maximum of perhaps 0.33 feet per year (see [Table 5](#) and [Figures 11 and 12](#)). There would not have been nearly enough time for a putative leachate plume to reach these wells from the disposal sites to the west.

Groundwater flow paths should provide a better indicator of source area. The closest upgradient potential sources from GWC-2 and GWC-3 would also have been some of the oldest at the facility, specifically the trenches in the Northeast quadrant (see [Figure 2](#)). Filling in that area began in 1984. GWC-2 is 600 feet away from the closest portion of trenches in the Northeast quadrant.

If we assumed that a plume did in fact reach GWC-2 by the year 2002, when metals concentrations began to rise, then 18 years would have elapsed from commencement of filling in the Northeast quadrant to detection at a downgradient well. Groundwater would have to flow at 33 feet per year to travel 600 feet in that period of time. We can be quite sure about the average groundwater gradient ([Figure 11](#)) because of the available database used to compute that, and the effective porosity value used is very typical for geologic materials of this sort. Therefore, the only variable remaining is hydraulic conductivity.

To travel at 33 feet per year, hydraulic conductivity would have to be about 2×10^{-3} cm/sec, a value that would be representative for a clean, moderately well-sorted sand. That is about two orders of magnitude higher than the hydraulic conductivities estimated for clayey fine sand near GWC-2.

Cross-sections E-E' ([Figure 8](#)) and B-B' ([Figure 5](#)) imply that some of the materials farther upgradient from GWC-2 and along a flowpath from the Northeast quadrant trenches might be silty sand. That sediment could have a hydraulic conductivity that was somewhat higher. Looking at site-specific test results in [Table 3](#), there were four valid slug tests for materials described as “silty fine sand” or “fine sand” (B-5, B-8, B-10 and B-22). All those values are in fairly close agreement, and their geometric mean is 2.7×10^{-4} cm/sec. This is still an order of magnitude too low to account for flow to GWC-2, and even that requires an assumption that all the intervening material was silty sand. Therefore, it appears very unlikely that any releases from the Northeast quadrant fill could have affected this well.

Because ash was not deposited in nearby areas, there is no leaching test data to compare against detections in these wells. However, metals detected in the leachate composite may be a partial approximation to leachate for the site as a whole. Analysis of the leachate composite sample included

antimony, arsenic, cadmium, chromium, lead, mercury, nickel and zinc, along with iron and pH. However, there are few opportunities for comparison against well data. Of those metals, only cadmium and nickel have appeared in GWC-2 often enough to be plotted for comparison.

Figure 30 shows cadmium and nickel data for the ditch and GWC-2. Figure 31 is the same data plotted to a vertical scale that shows well results more clearly, and the dashed lines represent detection limits for the well data.

The ditch water had large spikes in cadmium levels around mid-1995 and there were much smaller spikes as late as 1999, but concentrations were very low thereafter. At GWC-2, increases in cadmium above the detection limit began in April 2003 and there is no clear time relationship to ditch detections.

There were nickel spikes in ditch water between 1995 through 1997, but no comparable response in the well. Nickel levels spiking at about twice the level of those earlier ones were occasionally seen in the ditch from 2000 to 2004, but the monitoring frequency is only semi-annual at these points and it is difficult to know whether these are isolated occurrences or evidence of increased average nickel in leachate and runoff. There was a corresponding rise in nickel at GWC-2 that began in 2000. Whether the nickel in the ditch derives from waste or from changes in natural groundwater (via flow to the ditches) is unclear. For reasons already stated, it does not appear likely to come from waste anywhere near GWC-2.

Based on the above evidence, especially the absence of any source within a reasonable distance considering site properties, a landfill source for metals in the area near GWC-2 and GWC-3 is considered “unlikely”.

GWC-5

This well is located on the south side, across the ditch from the fill, and roughly opposite trench 45 as shown on Figure 2. In the Southeast quadrant, fill with MSW took place as far to the east as trench 43 at the most, and fill with ash was placed in trench 42. Groundwater flow toward GWC-5 from the vicinity of those trenches seems unlikely as shown in Figure 2. The most probable flow paths toward this well are from the

direction of trenches that never received waste before lining. Nevertheless, flow from the area of filled trenches might have occurred at times. As noted in the hydrogeologic Assessment, there is some possibility for groundwater flow to under-run the southern perimeter ditch at times after October 1998 or possibly earlier (Figure 21).

Any waste that occupied nearby trenches was present only between 1991 (at the earliest) and late 1996, when the waste was excavated as detailed in the Facility History (Appendix 4). Concentrations of beryllium have remained essentially uniform for about nine years (late 1996 to late 2005) after the waste was removed. The same is true for barium, although a high detection limit between early 1995 and early 2000 could have concealed some increases. These facts do not support landfilled waste as a source.

If waste constituents did reach this well, the maximum travel time would have been about six years, between initial disposal in 1991 and concentration increases in 1997. The nearest waste disposal in trench 43 would have been over 200 feet from GWC-5. As seen in Figures 11 and 12, the anticipated average flow velocity toward GWC-5 is 0.13 feet per year, with a maximum of 0.33 feet per year. In this case there is no apparent possibility of flow through any material except sandy clay. The estimated flow velocity using the highest observed gradient is 100 times too small to account for detections at this well after six years.

Beryllium also seems to be an unusual constituent for MSW. Apart from its use as an alloying element in manufacture of copper springs, spot welding electrodes and some aerospace components, beryllium salts were used in fluorescent lights prior to 1949. It would not appear to be common in non-industrial waste, but low concentrations of beryllium were seen in other site wells including AMW-1 (at higher concentrations), GWC-2 and GWC-3. From the evaluations above, these wells were classified as unlikely to have been affected by waste. Thus, low concentrations of beryllium are more likely to be derived from an environmental source.

Considering all the above factors including ground water flow and the duration of waste residence, a landfill source for metals in the area near GWC-5 is considered “unlikely”.

4.3 Acid Sulfate Soil as Potential Source

Based on the analyses presented in the preceding section, the waste unit at this facility was deemed unlikely to be the source of the noted impacts. Therefore, AEM conducted extensive research in an effort to find alternative sources for the noted impacts. Our research pointed to Acid Sulfate Soils as a potential source of the noted impacts at the facility. Field and laboratory tests were conducted to establish the presence of Acid Sulfate Soils at the facility. The presence of these soils was confirmed and is discussed in detail in [Appendix 8](#).

After establishing the presence of Acid Sulfate Soils at the facility and understanding the conditions whereby these soils can pose a threat to groundwater quality, it became clear that a greater understanding of historical climate conditions at the site would be required to fully evaluate the potential for these soils to cause the impacts noted. To this end, additional research was completed and the Assessment of *Climate and Acid Sulfate Conditions* attached as [Appendix 9](#) was completed.

In this section, the history and timing of increased metals concentrations in groundwater will be reviewed in connection with the rainfall, water level and pH data that appear to be related to acid sulfate soil conditions in an effort to assess the potential for Acid Sulfate Soils to be the source of impacts noted at the facility. As done in the previous section, the following discussion is grouped by the impacted locations.

4.3.1 GWC-11

[Figure 40](#) shows concentrations of barium and zinc in GWC-11 along with the pH levels for the period of monitoring. [Figure 41](#) shows concentrations of other selected metals in that well, along with pH and the “total precipitation” curve.

In these two charts and also subsequent charts for other wells, dashed lines represent detection limits as shown in [Appendix 10](#). The detection limits are not shown in the legend, but are color-coded to agree with the symbols and lines used with the concentrations of that metal. In situations where the detection limits are quite low or have remained fairly constant, there is no reason to include them in trend charts, but that has not been the case with some of the parameters during the past 13 years of monitoring at this site. Barium and zinc in particular have had fairly large detection limits in the past, and changes in the limits for arsenic and lead could affect interpretations at times.

Metals concentrations in the first few periods, during 1993 and 1994, could have been strongly affected by inadequate well development as previously noted and will not be considered for that reason.

As seen in [Figure 40](#), increased zinc levels were first apparent in October 1996; barium could have begun increasing at the time also, but would have been “masked” by the detection level. In any case, by April 1997 barium was about 1200 ppb and zinc had fallen below its 1000 ppb detection limit; zinc might have decreased similarly to barium through October 1998, but that is hidden by the detection level.

October 1996 also marked the first appearance of cadmium over the detection limit, and high concentrations of cobalt and nickel were seen. By April 1997, lead appeared briefly at around twice its detection limit, beryllium was barely above its limit and gradually decreased during the next few periods, cadmium continued a gradual rise and descent, and cobalt decreased below its limit, to reappear again later.

The total precipitation curve reached a low point in October 1996. The pH had begun to decrease, although there might have been an earlier pulse of low water levels and lower pH water that was missed between semi-annual samplings. The GWC-11 location is near to the borrow area and the steep drop in topography toward low-lying swamps. As discussed in the Hydrogeological Assessment ([Appendix 7](#)), these areas may have larger short-term swings in water level than other locations, so the beginnings of oxidation might have taken place several months prior.

If acid sulfate conditions were responsible, it may fairly be asked why there was no similar pH and metals response during later periods of low precipitation and low water levels, and why metals were not detected after the fairly brief April 2003 low pH event. While no simple answers present themselves, an assumption of waste as a source does not provide any convincing answers either. It is possible that pyrite in soils near this well had been largely depleted from earlier periods of low water level, and the sediments were largely leached of both pyrite and metals that could be mobilized because of the progressive expansion of the adjacent borrow area. In that case, the metals released from clays in 1996 and 1997 could have formed more stable complexes in the subsurface and, with less-frequent inputs of acid, remained out of solution. This is surmise only, but the initial rise in metals appears to fit the response expected for acid sulfate soils.

Acid sulfate soils as a source of metals at GWC-11 is classified as “probable” based on proximity to known areas of Acid Sulfate Soils, pH probe results that are similar to other locations where this source is very likely, the timing of metals increases in connection with lower precipitation in 1996-1997, and similarities to metals increases at other site wells. Although there are remaining uncertainties about the reasons for little or no response to later conditions favoring low groundwater levels, that alone does not appear to warrant a conservative classification of “possible” in light of the complex geochemistry that may be involved and the presence of partly leached soils nearby.

4.3.2 *AMW-1*

Figure 42 shows concentrations of barium and zinc in AMW-1 along with the pH levels, and Figure 43 shows concentrations of other selected metals and includes the “total precipitation” curve. The record begins in October 1998 after this well was installed, and levels of nickel were already elevated. Nickel concentrations declined through late 2002, and then began to rise again in parallel with cobalt, peaking around October 2003, declining again to a minimum in late 2004-early 2005, and rising again in October 2005.

The relationship to the total precipitation record and water levels in this well is complex. A full explanation would probably involve specific horizons with different degrees of pre-existing oxidation, metals complexes and flushing from those zones, and details of short-term water level changes. There is not enough information at hand to perform that analysis, but something is known about the soils surrounding this well.

On June 21, 2006, during the second of three site visits connected with the ASD investigations, excavation of the borrow area had proceeded to a point approximately 10 to 15 feet from the base of AMW-1. The well casing was then at the top of a promontory with a steep drop to saturated soils and standing water at the base. Soil was later placed against that exposure and the entire area regarded, but on that date it was possible to observe oxidized and leached soil with distinct orange and yellow horizons above saturated light gray sandy clay very close to the well. Sample AST-15 was collected from that saturated soil near the base AMW-1. The results of field peroxide oxidation tests on that sample are shown in Table 8, and were indicative of actual acid sulfate soil (AASS). The initial pH was somewhat low (4.72), the pH decrease on oxidation was small (to 4.03), but there was an unusual slow, delayed reaction resulting in a pH of 3.80 after eight hours. This suggests the presence of oxidized

sediments with metals complexes that break down over a prolonged period.

The pH record at this well is consistent with the above analysis, remaining around 4 despite changes in water level until a sharp decrease in October 2005. This may indicate that most pyrite has been oxidized, and pH will drop only when less-frequently exposed deeper horizons are exposed, even if briefly.

Based on the documented existence of AASS extremely close to this well, soil observations and testing, pH probe results and the apparent connection to environmental factors, acid sulfate soils as a source of metals at AMW-1 is regarded as “probable”.

4.3.3 GWC-2/GWC-3

Figures 44 and 45 present metals detections at GWC-2. The plot shows that a barium increase in April 2000 is not due solely to the change in detection level, but represents a sharp increase occurring around that date. That same date also marked the initial detections of cobalt and nickel, which began to increase sometime around April to October 2001 followed by cadmium in early 2003. Those increases are likely to be related to the prolonged period of decreased total precipitation in 2000-2002, and possibly short-term rises in water table as discussed previously with respect to pH trends.

The sudden appearance of vanadium at fairly high levels in October 2005, and the brief appearance of antimony in late 2003, remains unexplained.

Barium reached a peak in October 2001 and then began large fluctuations that seem to bear an inverse relationship to pH. Most barium in the subsurface occurs as barium sulfate, and possibly a combination of higher solubility at lower pH and levels of available sulfate ion is producing the large concentration swings.

Based on observations suggestive of acid sulfate conditions in soil near this well during the pH probe investigation, the pH probes themselves, and the relationship of precipitation and water levels to concentrations and pH history, acid sulfate soils as a source of metals at GWC-2 is regarded as “probable”. Contemporaneous changes in barium, cobalt and to some degree other metals, plus a comparable pH history, indicate that acid sulfate soils are similarly responsible for conditions at GWC-3.

4.3.4 GWC-5

Figures 46 and 47 present metals detections at GWC-5. In this case, the appearance of barium in late 2000 is caused by the change in detection limit, but actual levels might have remained fairly close to 100-200 ppb throughout the entire period of monitoring. The pH after October 1996 has stayed in the 4 to 5 range despite the fairly large changes in water level seen in Figure 34, and that could possibly suggest the presence of actual acid sulfate soils in which most of the pyrite has already been oxidized. The pH dropped abruptly in October 2005. The overall pH behavior seems very similar to that in AMW-1 where there are several lines of evidence pointing toward AASS. At GWC-5, the saturated sediments were described as gray to black in color, and the “black” descriptor is fairly uncommon in the monitoring well logs. As at the borrow area, a darker color may be characteristic of less-oxidized layers.

Concentrations of other detected metals (beryllium and nickel) have remained low and, most of the time, barely above the detection limits. Small nickel peaks in April 2000 and April 2001 seem to correlate with rises in water level after periods when levels were especially low (see Figure 34); as discussed previously, this could represent groundwater encountering strata where metals have been released through weathering.

The sudden appearance of vanadium in October 2005 is similar to that at GWC-2, and the cause is similarly unclear. In both instances the sharp drop in pH may be connected to vanadium appearance in some fashion.

Based on similarities to the pH response at AMW-1, the materials description, and a possible connection between water levels and metals detections, acid sulfate soils as a source of metals at GWC-5 is regarded as “probable”. In any case, there is little evidence of any meaningful increasing trends at this location.

4.4 Source Analysis Summary

The evaluations of potential sources for metals detections in areas of concern are summarized in the following table:

	GWC-11	AMW-1	GWC-2/ GWC-3	GWC-5
Landfill Source	Possible	Unlikely	Unlikely	Unlikely
Acid Sulfate Source	Probable	Probable	Probable	Probable

On the balance of the evidence, a natural source for metals, associated with acid sulfate soils in various parts of the site, is considered the most likely explanation for the existing impacts at this facility. Landfill waste as a source would be difficult to explain in terms of the site characteristics and waste disposal history at these locations, and the detection histories are very similar to those in areas where acid sulfate conditions are strongly indicated.

5.0 CONCLUSIONS

1. The Dean Forest Road facility is underlain by sediments composed of sand, clay and silt that appear to have formed in coastal lagoons and associated barrier islands more than 130,000 years ago. Features of that depositional environment are compatible with biogenic pyrite fixation in fine-grained sediments. Those sediments can later develop acid sulfate soil conditions when saturated sediments are exposed to air. The resultant acidity can have effects on groundwater, including the release of metals adsorbed on clay.
2. Acid sulfate soils are better known and have been more thoroughly studied in other parts of the world, but have been increasingly recognized at various sites along the eastern US coastal plain, principally within the past few years.
3. Acid sulfate soils are present immediately west of the landfill as confirmed by observations at exposures in an area undergoing dewatering, field and bench tests, and laboratory testing of soil samples. Concentrations of inorganic sulfur, primarily pyrite, are high in comparison to action levels governing dewatering or other disturbance in areas with established standards.
4. Boring and well logs indicate a generally similar suite of sediments throughout the property. This is consistent with the transgressive marine depositional conditions that appear to be responsible for the site sediments. Those observations imply that acid sulfate soils could develop at many locations throughout the site.
5. Field observations near well GWC-2, during pH probe investigations, suggest that acid sulfate soils are present on the east side of the landfill in the immediate vicinity of that well. The pH probe investigations mapped vertical and horizontal changes of pH near areas of concern, and the results also implied that acid sulfate conditions are present.
6. Low-pH conditions are continually or intermittently present at many downgradient monitoring wells. Leachate and runoff to perimeter ditches appears to be near neutral and cannot be responsible for the depressed pH in the wells.
7. Groundwater flow at the site is radial, moving away from the landfill toward adjacent swampy lowlands that are probable sites of ultimate groundwater discharge.
8. Based on geologic cross-sections and an extensive review of formation tests, soil properties and water level history, estimated groundwater flow velocities are very low near the perimeter of the landfill. The average estimated flow velocity in any direction is less than one-quarter foot per year, and the

maximum velocity under the highest gradient conditions is well under one foot per year. This is a consequence of low groundwater gradients and the low permeability of most site materials.

9. Groundwater does not flow downward from the shallow saturated zone for lengthy periods of time on the northern and eastern sides of the landfill. Small downward or upward gradients may exist at times. That conclusion is based on analysis of water levels for three pairs of wells. Because no well pairs exist elsewhere, it is not possible to assess the likelihood of downward flow in other parts of the landfill, but there is no reason to believe this is taking place.
10. Groundwater flowing away from the landfill appears to have a potential for discharging to the perimeter ditch on the northern and eastern sides of the landfill. A pressure transducer study demonstrated very similar response to an intense storm in ditch water and in a nearby well (GWC-2), indicating that groundwater is hydraulically connected to the ditch in that area. Groundwater probably discharged to the ditch on the western side of the landfill until recently, when dewatering in the borrow area and a continued decline in water levels brought water levels below the ditch elevation. At places on the southern side of the landfill, groundwater may be intercepted by the ditch or may under-run the ditch at various times.
11. Toxicity Characteristic Leaching Procedure (TCLP) analyses of older, untreated waste combustion ash indicates a potential for release of barium, cadmium and lead. However, testing of subgrade soils after that ash was removed indicates that those metals were not released to the soil column, and that ash leachate would be unlikely to have more severe metals releases than municipal solid waste on average. The disparity is due to the aggressive nature of the TCLP test, which assumes co-disposal with organic waste under anaerobic conditions, which did not apply to ash disposal at this facility.
12. Acid sulfate soils are the most likely cause of metals detections in areas of concern: near wells GWC-11 and AMW-1 on the western side of the landfill, in the vicinity of wells GWC-2/GWC-3 on the eastern side, and near GWC-5 on the southern side. These conclusions are based on a systematic analysis of the landfill and of acid sulfate soils as potential sources for metals.

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