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STATE OF ILLINOIS
POLLUTION CONTROL BOARD

TESTIMONY OF
MR. DAN OMAN
ON BEHALF OF
THE ILLINOIS CAST METALS ASSOCIATION
THE ILLINOIS STEEL GROUP

R90-26,
SOLID WASTE RULES FOR
THE ILLINOIS FOUNDRY AND
STEEL INDUSTRIES

MAY 29, 1991

LANDFILL DESIGN STANDARDS

I. <u>LINERS</u>

One of the objectives of proper landfill design is to protect ground water quality. This can be achieved by siting landfills in suitable geologic (e.g., clay-rich) environments that naturally limit the migration of potential contaminants into the ground water. However, in many areas of the country, this type of geologic environment is not locally available. In these instances, it is common practice to construct some form of liner system (i.e., clay and/or geomembranes) to provide the necessary protection of ground water.

Typically, liners are designed and constructed so as to achieve a maximum hydraulic conductivity (e.g., 1×10^{-7} cm/sec). The minimum liner thickness necessary to achieve this design hydraulic conductivity is often debated.

According to Dr. David E. Daniel of the University of Texas (April 1990):

> "With sound construction practices, one should be able to construct a soil liner that has an *in situ* hydraulic conductivity that is less than or equal to 1×10^{-7} cm/sec, if the soil liner is at least 2.0 feet thick."

Daniel (April 1990) went on to conclude in the same study:

> "Increasing the thickness of a soil liner from 2.0 to 3.0 feet will likely lower the hydraulic conductivity of the liner by a factor of 2, increase slightly the already high probability that hydraulic conductivity will be less than or equal to 1×10^{-7} cm/sec, offer some buffer from possible damage of the liner due to desiccation, frost action, or settlement, and generally add a "factor of safety" to the design. Perhaps the answer to the question of whether a 3-foot-thick liner is needed rather than a 2-foot-thick liner should hinge on other issues, such as whether the soil liner will be used with a flexible-membrane liner to form a composite (in which case, the extra foot of soil liner thickness probably provides almost no measurable improvement in performance of the composite liner)."

Dr. Daniel based these two conclusions on a careful review of the literature, his files, and information recently compiled by one of his graduate students. Basically, Dr. Daniel compared liner thickness to *in situ* hydraulic conductivity for 23 different soil liners, which were built with what he has termed as "good to excellent construction practice" and that had a thickness of 2 feet or more. Of the 23 liners, 22 of them, or 96 percent, had hydraulic conductivities $\leq 1 \times 10^{-7}$ cm/sec. The one soil liner which exceeded this limit had a hydraulic conductivity of 2 x 10^{-7} cm/sec. Dr. Daniel hypothesized that longer term testing may have resulted in a hydraulic conductivity value for this site below 1 x 10^{-7} cm/sec.

In another study, Benson and Daniel (November 1990) applied two models that simulate the flow of water in saturated, compacted soil liners. The purpose of this effort was to answer the question "How thick should a compacted soil liner be?" As a result of this work, Benson and Daniel concluded the following:

"3. The field data and models show that soil liners that are 15 to 30 cm thick (1 or 2 lifts) tend to be much more permeable than liners that are 60 to 90 cm thick (4 to 6 lifts). **Little reduction in hydraulic conductivity is achieved when the thickness is beyond 60 to 90 cm (4 to 6) lifts.** (emphasis added)

4. If at least 4 lifts are used, the degree of bonding between lifts, i.e., the degree to which zones and high horizontal hydraulic conductivity at lift interfaces are eliminated, is far more important than the number of lifts. Stated in practical terms, the overall hydraulic conductivity of a well-built liner composed of 4 lifts (60 cm) will be far lower than the hydraulic conductivity of a poorly-built liner containing many more lifts. Adding more lifts (60 - 90 cm) will not ameliorate the problems left from poor construction (poor bonding between lifts, or high mean hydraulic conductivity within a lift).

5. A reasonable minimum thickness for low-hydraulic-conductivity, compacted soil liners is 60 to 90 cm (4 to 6 lifts)."

The research indicates that a minimum liner thickness of 60 cm (2 ft) should be sufficient to provide a maximum permeability of 1 x 10^{-7} cm/sec. Thus, the 90-cm (3-foot) liner thickness proposed in R 90-26 Section 811.1106(d)(l) is on the upper end of the minimum thickness suggested by Benson and Daniel.

The research also indicates that quality control during construction can be <u>more</u> important than additional thickness in providing a low–hydraulic conductivity liner. The construction quality control regulations contained in R 88-7 should help to ensure that clay liner construction is of suitable quality to achieve the desired maximum hydraulic conductivity.

In addition, the wastes generated by the foundry and steel industries are of a physical nature that minimizes the potential for the wastes to puncture the clay liner. This is contrasted with the physical characteristics and heterogeneous nature of municipal solid waste which does have the potential to cause significant damage to the liner during the placement and compaction process.

Although formal liner compatibility studies have not been conducted, steel and foundry waste leachate typically does not contain organic solvents at concentrations high enough to materially degrade clay liner performance. In addition, to my knowledge, there have been no documented cases at foundry or steel waste landfills where clay liner performance has been materially affected by exposure to leachate. Therefore, there is no need to increase liner thickness to allow for physical or chemical damage.

II. FINAL COVER

The basic purpose of a landfill final cover is to minimize infiltration of water into the landfill. Other purposes for a final cover include the following:

- Minimizing the potential for erosion of waste materials.
- Prevention of direct contact with the waste.

To properly minimize infiltration, final cover systems of landfills typically involve some form of low-permeability layer constructed of materials such as clay soil. Ideally, a low-permeability layer designed as part of the final cover system should achieve a hydraulic conductivity which does not exceed the hydraulic conductivity of the liner system. In this way, the final cover system will not allow a significant amount of infiltration into the site which would contribute to a buildup of leachate head on the liner system.

The thickness of the low-permeability layer necessary to achieve this maximum permeability should be no different than the thickness required for a liner system. This assumes that the material (i.e., waste) underlying the low-permeability layer presents a very stable foundation for compaction purposes. Based upon the research mentioned previously concerning liner thickness, a low-permeability layer of at least 2.0 feet in thickness should be suitable for achieving an effective hydraulic conductivity of 1 x 10^{-7} cm/sec.

In addition, the physical characteristics of the foundry/steel wastes make them ideally suited as a base for construction of the low-permeability layer. As placed, these materials are very stable and undergo very minimal settling. This is in sharp contrast to the settlement which can occur at municipal solid waste (MSW) landfills. Also, installation of a low-permeability layer on top of MSW can be difficult because of the "spongy" nature of the waste.

Another component of the final cover system at a landfill is the final protective layer. The purpose of this layer is to protect the low-permeability layer from frost and root penetration. Frost penetration is typically a function of climate and soil texture. Soils which are granular in nature (e.g., sandy soils) will usually freeze deeper than heavier, wetter clay and silt loam soils. Maximum mean frost depths in Illinois typically do not exceed 16 inches with the exception of the extreme northwestern portions of the state. The final protective layer must be at least this thick to minimize the potential for exposure of the low-permeability layer to freeze-thaw cycles.

A study conducted by Grefe, Huebner, and Gordon (September 1987) investigated the root penetration depth of cover materials at 22 landfills in Wisconsin. Their research indicated that the maximum root penetration was 18 inches. This penetration depth was measured at five locations out of a total of 77 samples. Therefore, the proposed thickness for the final protective layer of 18 inches should be adequate to minimize the potential for root penetration.

III. LEACHATE COLLECTION

The purpose of a leachate collection system is to collect and convey leachate which is trapped above the landfill liner. Without a leachate collection system, leachate would simply pool above the liner until enough hydraulic head is built up to force the leachate to flow through the liner. The leachate head may eventually reach an equilibrium level at which the flow through the liner is equal to the rate of infiltration through the final cover. If the rate of infiltration through the final cover exceeds the rate of flow through the liner, the leachate head will continue to rise until the equilibrium level is achieved.

Leachate from foundry and steel wastes is generated as infiltration percolates through the waste. Because these wastes are mineral in nature, leachate is not generated as a result of biological degradation of the material. Instead, the soluble constituents associated with these wastes are leached out from the waste as water percolates through the fill. This is in sharp contrast with MSW which generates leachate as the waste biologically decomposes within the fill.

Because of the nature and characteristics of the foundry and steel wastes, and the lack of significant environmental impacts associated with existing disposal facilities which contain these wastes, a leachate collection system designed for a new monofill should not have to function like a conventional MSW landfill leachate collection system. Its basic purpose is to provide a mechanism for periodic leachate head reduction in those instances when leachate levels interfere with site operation. Stated another way, the leachate collection system is basically a back-up system that can be operated to reduce leachate head levels only when they become excessive enough to interfere with landfill operations.

Following closure, the leachate heads in the monofill should be monitored to determine whether or not an equilibrium level has been achieved. If leachate levels continue to rise to a point where the integrity of the liner is jeopardized or leachate seeps are possible, the leachate collection system should be operated to reduce the leachate head. Continued periodic operation of the leachate collection system, coupled with monitoring of leachate quality, will be conducted during the post-closure period. An assessment would then be made as to whether or not leachate collection could be discontinued.

IV. DESIGN PERIOD

The purpose of a design period for a landfill site is to specify the minimum period of time that items of construction such as the liner and leachate collection system, must function. The design period specified by the current rules for chemical and putrescible landfills is the operating life of the landfill plus 30 years. This design life is based on the fact that MSW can undergo decomposition for a significant period of time following initial disposal. The Board has realized that this degradation of MSW can be enhanced through scheduling or leachate recycle within the fill. In these cases, the Board has recommended that the design period be reduced to the operating life of the facility plus 20 years.

Steel and foundry industry wastes do not undergo the type of biological decomposition that MSW undergoes. In fact, these materials are typically very inert and behave much like soil materials in the landfill. They compact well and are not susceptible to the type of settlement experienced at MSW landfills. Leachate studies have shown that these wastes typically release those chemical constituents not limited by solubility very early in the leaching process. In other words, leachate quality will tend to remain very consistent with time during the operating life of the site. Once the site is closed, these wastes will tend to stabilize fairly soon, and a reduction in leachate strength will occur. Also, the strength of leachate from these wastes is far less than the strength of leachate from MSW. Therefore, a design period that includes the operating life of the site plus 20 years should be adequate for the design of liners and leachate collection systems for landfills that receive these wastes.

LITERATURE CITED

Daniel, David E. April 9, 1990. Note on Thickness of Compacted Soil Liners. In Seminars—Design and Construction of RCRA/CERCLA Final Covers. Appendix A, p. A-1 - A-18. United States Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Benson, Craig H., and David E. Daniel. November 1990. Minimum Thickness of Compacted Soil Liners: II-Analysis and Case Histories. Submitted to: Journal of Geotechnical Engineering. American Society of Civil Engineers. New York.

Grefe, Robert P., Paul M. Huebner, and Mark E. Gordon. September 1987. Multilayered Cover Design and Application to Wisconsin Landfills. In Proceedings of the Tenth Annual Madison Waste Conference. Department of Engineering Professional Development, University of Wisconsin - Madison.

EXHIBIT

A

DANIEL E. OMAN, P.E.
Vice President, Northern Region

ROLE AT RMT:

As Vice President of RMT's Northern Region, develops and directs the region's solid waste, environmental management, industrial hygiene, and asbestos management programs.

- Provides senior QA/QC for the following:
  - Landfill siting, design, and construction projects.
  - Waste characterization and waste minimization studies.
  - Hazardous waste management projects.
  - Site assessments and preacquisition and compliance audits.
  - Remedial action projects.
- Manages large projects for industrial clients.

REPRESENTATIVE PROJECTS:

Manages a number of large projects for major industrial clients involving solid and hazardous waste management. RMT project work includes the following:

- *Industrial Landfills* - Managed several industrial landfill projects for major clients regarding all facets of landfill siting, design, and construction.
- *Remedial Action Alternatives Development* - Conducted options analyses regarding the development of remedial action alternatives, weighing the technical, regulatory, and economic aspects of each alternative.
- *RCRA Closure* - Developed closure strategies and determined "how clean is clean" for closure of hazardous waste surface impoundments, waste piles and lagoons.
- *Hazardous Waste Treatment* - Managed several projects for industrial clients to render hazardous waste nonhazardous.
- *Hazardous Waste Minimization* - Implemented a charge modification program to reduce lead concentration in sludge, and developed alternatives for management and disposal of hazardous and nonhazardous sludges.
- *Waste Characterization* - Conducted numerous waste characterization studies for industrial clients in more than ten states.

EDUCATION:

M.S., Civil and Environmental Engineering, University of Wisconsin-Madison. *Thesis: A Comparison Between Two Laboratory Batch Leaching Procedures and In-Field Leachate Quality at Foundry Waste Landfills.*

B.S., Civil Engineering, University of Delaware

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PROFESSIONAL REGISTRATIONS:

Registered Professional Engineer

PROFESSIONAL AFFILIATIONS:

Member of:

- American Society of Civil Engineers
- Technical Association of the Pulp & Paper Industry
- The Hazardous Materials Control Research Institute

PRESENTATIONS AND PUBLICATIONS:

1991 TAPPI Environmental Conference, April 7-10, 1991, San Antonio, Texas.
- Successful Landfill Siting: Location is the Key
- Chemical Compatibility Testing of Geomembranes and Piping Materials Exposed to Pulp & Paper Mill landfill Leachate.

American Foundrymen's Society, 40th Annual Northwest Regional Conference, February 28, March 1-2, 1991, Seattle, Washington.
- Measuring for Compliance
- Hazardous Waste Treatment Options

MO-KAN Chapter of the American Foundrymen's Society, February 21, 1991, Kansas City, Missouri, TCLP, and the Foundry Industry.

Foundry Waste Disposal in Ohio Seminar, November 3-10, 1990, Cleveland & Columbus, Ohio.
- Overview of Proposed Rules
- A Broader Perspective

Northwestern Pennsylvania Chapter of the American Foundrymen's Society, April 17, 1990, Erie, Pennsylvania. Surviving the May 3, 1990, Land Ban.

Institute of Boiler & Radiator Manufacturers, May 10, 1989. Waupaca, Wisconsin, Waste Management in the Foundry Industry: Where Are We Headed?

1989 TAPPI Environmental Conference, April 16-19, 1989, Orlando, Florida, Leachate Quality from a Pulp & Paper Mill Ash Landfill.

Western New York Chapter of The American Foundrymen's Society, March 3, 1989, Buffalo, New York, Managing Your Hazardous Waste.

American Foundrymen's Society Total Environment Conference, August 10-11, 1988, Milwaukee, Wisconsin, The Changing Solid Waste Management Regulatory Environment: A Consultant's Perspective.

American Paper Institute, Tissue Division, Small Mills Program, June 6-8, 1988, Appleton, Wisconsin, Landfill Siting Strategies.

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National Research Concil Committee to Evaluate Mass Balance Information for Facilities Handling Toxic Substances Workshop, March 24-26, 1988, National Acedemy of Sciences, Washington, D.C. The Application of Mass Balance Techniques in the Foundry Industry.

American Paper Institute, Tissue Division, Technical Committee, March 7, 1988, New York, New York, The Changing Solid Waste Management Environment: A Consultant's Perspective.

A Fast-Track Approach Through the Design and Permit Phases for an Ash Landfill, *TAPPI Journal*, August 1988.

Waste Minimization in the Foundry Industry, *Air Pollution Control Association Journal*, July 1988.

Technical Conference, Succeeding at Waste Minimization, University of Wisconsin-Extension, April 25-27, 1988. Case Studies in Foundry Waste Minimization.

The 1988 TAPPI Environmental Conference, A Fast-Track Approach to Design and Permitting of a Paper Mill Ash Landfill.

Central Lake States Regional Meeting of the National Council of the Paper Industry for Air and Stream Improvements, September 18, 1986, Design and Operation of Pulp & Papermill Sludge Landfills.

Using Foundry Sand in Highway Construction, April 23-24, 1986. Important Considerations When Characterizing Foundry Wastes.

Effective Management of Industrial Wastes, University of Minnesota, June 18, 1985. Effective Management of Industrial Wastes.

American Foundrymen's Society - Wisconsin Chapter Meeting, March 18, 1985. Waste Characterization and Analysis: A Key to Waste Management Alternatives.

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TESTIMONY OF
MR. CHRIS PETERS
ON BEHALF OF
THE ILLINOIS CAST METALS ASSOCIATION
THE ILLINOIS STEEL GROUP

R90-26,
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GROUND WATER MODELING AND GROUND WATER IMPACT ASSESSMENT

I. SUMMARY

In the following sections, we provide a discussion on the shortcomings of ground water contaminant transport modeling for use in assessing potential ground water impacts adjacent to landfills. While we acknowledge that there is some benefit to contaminant modeling, particularly when comparing the attributes of two or more different sites, we do not believe that this type of modeling is necessary or even appropriate for foundry or steel waste landfills. Other testimony provided by Dr. Stanforth demonstrates the low-leaching potential of this waste type, particularly in a monofill environment. We do not believe that the considerable expense of performing a contaminant transport model is warranted given the low leachability and minimal ground water impacts expected as a result of foundry and steel disposal operations. A more simplified approach to assessing ground water impacts, using the federal maximum contaminant levels (MCLs), is proposed as an alternative.

II. R88-7 GROUND WATER MODELING REQUIREMENTS

R88-7 requires that a ground water contaminant transport (GCT) model be used to perform a ground water impact assessment. A GCT model must be used to predict the concentration of all leachate constituents in the zone of attenuation surrounding the landfill (the zone of attenuation extends 100 feet from the edge of the landfill or to the property boundary, whichever is less). The facility must be designed such that the ground water quality standards specified in 35 Illinois Administrative Code 811.320 are not exceeded outside the zone of attenuation within 100 years of closure.

The model is also used to develop the maximum allowable predicted concentrations (MAPCs) which apply to all monitoring points within the zone of attenuation. The same calculations, data, and assumptions used in the ground water impact assessment must be used to predict the concentrations through time of all constituents at various locations within the zone of attenuation.

The monitoring points at which the MAPCs apply must be located as close to the landfill as possible, within half the distance from the edge of the landfill to edge of the zone of attenuation. At least one downgradient monitoring well must be located at the edge of the zone of attenuation, i.e., at the compliance boundary. The well will be used to monitor any statistically significant increase in concentration of any constituent.

If the concentration increase is confirmed, ground water assessment monitoring must be done to determine if the landfill is the source of the ground water impact. If the landfill is determined to be the source of the ground water impact and constituents at concentrations exceeding ground water quality standards are found at or beyond the zone of attenuation, a ground water impact assessment must be performed to determine the extent of ground water contamination. The assessment must evaluate the impact on ground water if the facility continues to accept waste and must determine what remedial action is necessary. Similarly, if the landfill is determined to be the source of ground water impact, and MAPCs are exceeded within the zone of attenuation, then a ground water impact assessment must be completed. The assessment is made using a recalibrated GCT model to determine if remedial action is necessary.

III. DESCRIPTION OF MAJOR GROUND WATER MODELS AVAILABLE

In this section, we will provide a summary of ground water contaminant transport models and the assumptions involved in their development. Our focus will be on numerical models since numerical models are required by R88-7. Dozens of contaminant transport models are on the market today. In an overview and status report on ground water modeling, van der Heijde et al. (1988) list 73 contaminant transport models. Our purpose here is not to do an exhaustive comparison of these, but to discuss the models' similarities, the model assumptions that can affect the results, and the relative costs for performing these models.

Most industrial waste leachates will produce concentrations of chemical species that are low enough so that the specific weight of the leachate will not be substantially different than that of the receiving ground water. In these cases, a contaminant transport model is really two submodels: a flow submodel used to predict the distribution of hydraulic head and estimate ground water fluxes, and a quality submodel to predict concentrations of chemical species at different points within the flow domain.

Model Inputs

The least complicated contaminant transport models are those used to predict concentrations of conservative, or non-reactive, species such as chloride. These models generally only consider advective ground water transport, or that which is predicted by the flow model, coupled with the effect of dispersion or diffusion, which results in spreading of a contaminant transverse and parallel to the ground water flow direction. While these models do not consider the complexities introduced by chemical reactions, even these models are not totally reliable because of the difficulty in defining dispersivity. This will be discussed later in this presentation.

To simulate concentrations of nonconservative chemical species, the various chemical and biological processes that occur within the soil media must be considered. Those likely to be significant for industrial wastes include the following:

- ***Adsorption*** - a process in which dissolved chemicals in the ground water become attached to soil particles and/or organic matter, and are removed from the dissolved phase.

- ***Transformation/Degradation*** - processes that determine the fate and persistence of chemical species in the ground water environment. These processes include biotransformation, chemical hydrolysis, oxidation/reduction, precipitation/dissolution, and ion speciation/complexation. In the contaminant transport equation, these processes are usually combined into one reaction term (van der Heijde et al., 1988). The reaction rates depend on several variables, including organic matter and temperature.

Van der Heijde et al. (1988) provide a comparison of the features of available contaminant transport models. Most of the 73 numerical models compared in their study incorporate the processes described above, and several incorporate additional processes.

__Limitations of Contaminant Transport Models__

As is the case with ground water flow models, the results of contaminant transport models are affected by the assumptions used. Unfortunately, these assumptions can be highly variable and inaccurate. For example, dispersivity values have been shown to be a function of scale and to increase with distance from the source (Gelhar et al., 1979). However, even within the 100-foot attenuation zone distance, as presented in R88-7, longitudinal dispersivity may vary over two orders of magnitude (Anderson, 1984). Dispersion in porous media is difficult to accurately estimate because of small-scale heterogeneities, such as sand and gravel lenses, that may occur even within deposits that are considered relatively uniform. Uncertainty of this magnitude makes prediction breakthrough times of various chemical species at a downgradient point very difficult.

Adsorption terms may also not be reliable for a few reasons. First, as discussed in the appendix to Illinois Administrative Code Section 811.317 (ground water impact assessment), laboratory techniques are used to generate distribution coefficient (K_d) which is used to predict the degree to which nonconservative species will be retarded with respect to the ground water flow. Typically, K_d values are generated for individual species through batch laboratory tests using solutions of different concentrations of the chemical constituent with different volumes of soil. While we agree with the contention in the appendix for 811.317 that the procedure is reproducible, the procedure is performed in a laboratory, using individual species in solution. Leachate is a complex mixture of chemicals, and the interaction between the chemicals can affect the adsorption behavior of the solid. In most cases, actual leachate samples from the field will not be available prior to the time the landfill is built. Furthermore, it is difficult to accurately simulate field conditions in the laboratory. Bond and Hwang (1988) state that "one of the biggest problems with simulating natural ground water conditions in the laboratory is properly representing the geologic media (i.e., layering, heterogeneity, etc.)." Adsorption would not generally be considered a problem in coarser grained soils where this process is not as significant in affecting ground water chemistry.

Chemical reactions are even more difficult to accurately predict. Several reactions can occur within the ground water flow system. These include oxidation and reduction (redox), precipitation/dissolution, and ion speciation/complexation reactions. Oxidation/reduction reactions are extremely important in ground water systems in that they control solubility of minerals, affect soil adsorption processes, and control contaminant migration, particularly of metal contaminants likely to be found in industrial waste. Several variables control redox reactions, including oxygen content of the recharged ground water, distribution of organic matter and redox buffers, and the circulation rate of ground water (NWWA, 1990). This requires measurement of Eh (redox potential, or essentially a measure of the oxidizing or reducing conditions within the sample), dissolved oxygen, and organic carbon content, which are atypical parameters in ground water monitoring programs. Eh is also sensitive to changes in atmospheric pressure (Ault, 1982); thus, reproducibility may be difficult.

These reactions also assume that chemical equilibrium conditions exist within the ground water flow system. This assumption is questionable, particularly with a system of small scale, such as that immediately downgradient of a landfill, and a system that has been stressed by a contaminant release.

Not discussed here, although equally as important, are the assumptions used for the ground water flow model inputs. Heterogeneities in the field can significantly alter the assumed model inputs and therefore cause predictive errors.

The cost for developing the data and running a contaminant transport model can vary widely. The calibration of the model requires the collection of background ground water chemistry of an extended list of parameters from several monitoring wells surrounding the landfill. To account for seasonal variability in constituent concentrations, it would be desirable to collect up to four rounds of data from these wells. The cost of collecting and analyzing the ground water samples varies depending on the size of the landfill and the number of chemical parameters, but $10,000 to $30,000 may be a reasonable estimate. The cost for running the model can vary also, depending on the number of factors used to predict contaminant concentrations, and the quality assurance requirements dictated by the regulatory agency reviewing the model. A reasonable estimate for this cost may be $15,000 to $30,000. These estimates do not include the cost of providing site-specific geochemical data, such as generation of K_d values for various parameters.

Given the evidence presented here, it would seem illogical to base remediation and enforcement actions on a high-cost model that is only conceptual, when a simpler, lower-cost alternative would suffice to assess ground water impacts from steel and foundry waste monofills covered by Section 817. This alternative is discussed in Section IV.

IV. ALTERNATIVE APPROACH

While we acknowledge that there may be some benefit to the use of contaminant transport models, we do not believe that their use for monofills containing foundry and steel industry wastes is justified. The wastes in steel and foundry monofills are a known commodity, as are the expected constituents of the leachate. If those constituents are affecting ground water quality, there is no need for an elaborate, expensive model to predict concentrations. Instead, we would propose an alternative approach based on the use of the federal Maximum Contaminant Levels (MCLs) as guidelines. Our proposal states that MCLs shall not be exceeded at the compliance boundary and beyond. Concentrations inside the compliance boundary in excess of the MCL would technically not be in violation of water quality standards. However, our proposal also includes a proposal for defining a concentration level that is some percentage of the MCL that would apply at any location where ground water is monitored (inside or outside the compliance boundary), and that, if exceeded, would trigger a response on the part of the facility. This response would be a written notice of exceedance of this lower concentration standard, and possibly a requirement for further action, depending on the circumstances. This further action may be more frequent monitoring, additional parameters, or some other appropriate action. For the purpose of consistency with R88-7, we have designated this lower concentration standard as the MAPC.

Use of this MAPC would serve as an "early warning system" for potential ground water quality impacts. This would also allow flexibility in the location of monitoring wells, by not necessarily requiring wells to be placed halfway between the waste boundary and the compliance boundary. This is not always practical or possible adjacent to a landfill where access roads, sedimentation basins, and perimeter surface water drainage swales may be located.

These lower concentration standards have been successfully used in Wisconsin since 1985, when NR140 (Exhibit A), the Wisconsin Administrative Code for ground water quality, was adopted. This lower standard, called a Preventive Action Limit or PAL, is typically 10 to 20 percent of the Enforcement Standard concentration (which, for inorganic parameters is nearly always equivalent to the federal MCL). We would suggest that a concentration of 10 percent of the MCL be adopted. In the case where the background concentration of a constituent may be above this MAPC, the percentage would be added to the background concentration to determine the MAPC.

We would also suggest that s ndards be employed for parameters tha have a federal secondary maximum contaminant leve MCL) (for example, chloride and manganese), and any parameters indicative of waste lea ate migration (indicator parameters) that do not have a standard associated with them. A ocedure similar to that of the MCLs would be used for the SMCLs. However, since the SMCLs are not health related, we suggest using an MAPC of 50 percent of the SMCL to trigger a response. Again, the SMCL could be exceeded inside the zone of attenuation. For indicator parameters, a background concentration would be established based on pre-disposal water quality sampling. A response would be triggered based on a statistical exceedance of the established background concentration.

While this procedure is obviously less complicated than developing MAPCs as required by R88-7, we believe that the effort to develop them by that procedure is not warranted for steel and foundry industry monofills.

REFERENCES

Anderson, M.P. 1984. Movement of contaminants in ground water: ground water transport-advection and dispersion. In Ground Water Contamination National Academy Press, 1984.

Ault, K. 1982. A review of geochemical computer models for equilibrium calculations with applications in natural aqueous systems. International Ground Water Modeling Center, Holcomb Research Institute, Butler University.

Bond, F., and G. Hwang. 1988. Selection criteria for mathematical models used in exposure assessment ground water models. USEPA Report 600/8-88/075.

Gelhar, L.W., A.L. Gutjahr, and R.L. Naff. 1979. Stochastic analysis of macro dispersion in a stratified aquifer. Water Resources Research, V. 15, p. 1387-1397.

National Water Well Association. 1990. Introduction to ground water geochemistry. Short Course, Toronto, Ontario, September 17-19, 1990.

Van der Heijde, P.K.M., A.I. El-Kadi, and S.A. Williams. 1988. Ground water modeling: an overview and status report. International Ground Water Modeling Center, Holcomb Research Institute, Butler University, GWMI 88-10.

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STATE OF ILLINOIS
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TESTIMONY OF ROBERT R. STANFORTH
ON BEHALF OF THE ILLINOIS CAST METALS ASSOCIATION

1. **INTRODUCTION**

My name is Robert Stanforth, and I am testifying on behalf of the Illinois Cast Metals Association and the Illinois Steel Group regarding the proposed steel and iron foundry amendments to the landfill regulations (Parts 810-815). My testimony pertains to the leaching procedure used for evaluating the foundry wastes, and the proposed numerical criteria used to classify the wastes.

2. **PROFESSIONAL BACKGROUND AND EXPERIENCE**

I am a water chemist, with MS and PhD degrees from the University of Wisconsin-Madison in water chemistry. I am currently employed as a Senior Applied chemist at RMT, Inc. of Madison, Wisconsin, an environmental consulting firm. I have worked at RMT for seven years. I have also taught environmental chemistry at the National University of Malaysia for three years.

My areas of expertise at RMT is in the evaluation and treatment of foundry and steel mill wastes, both to determine their potential impact on groundwater and to find ways of rendering the wastes nonhazardous and nonleachable. While at the University of Wisconsin, I worked for a year on a background study for the development of a standardized leaching test for the US Environmental Protection Agency and on an American Foundrymen's Society study on the leaching characteristics of foundry wastes. In addition, after joining RMT I participated in an evaluation of the impact of ferrous foundry landfills on groundwater. All of these studies are germane to the proposed regulations being discussed today.

3. **PROPOSED LEACHING PROCEDURE**

The purpose of the proposed leaching test is to provide a rapid and simple means of screening wastes to determine which might leach undesirable amounts of constituents into the environment. Constituents that leach in a leaching test will likely also leach after disposal, although not necessarily at the same concentration as found in the leaching test. By measuring what leaches in the leach test and comparing the results of parameters of concern with a set of criteria, we can evaluate which wastes may leach undesirable concentrations of constituents and therefore should be land disposed in less restrictive conditions. However, any leaching test provides a simplistic model of what actually occurs in a landfill, and the results that are generated are several steps removed from the concentrations likely to be seen in a landfill leachate. Concentrations of some parameters in landfill leachate will vary dramatically over time. Some parameters will be rapidly leached from the waste at high concentration, with low residual concentrations after the initial flush of material has passed through. The concentrations of other parameters may be highly dependant on the composition of the leaching solution or the oxidation-reduction status of the waste material. Using the results of a single leaching test conducted under a specified set of conditions is a simplistic means of predicting the leaching character of the waste.

However, the need for a simple screening tool for estimating the leaching potential of a waste overrides the technical difficulties of extrapolating the leach test results tofield leaching situations. It is important to realize, however, that the leaching test is simply an easily applied tool for evaluating leaching potential from a waste, and does not accurately model all of the leaching conditions that will be found in all landfills. The leach tests in short, are a convenient regulatory tool for classifying wastes and can provide useful information on the leaching character of the wastes under a given set of conditions.

3.1 Description of test.

The leaching test proposed for use in classifying foundry and steel wastes in the ASTM Standard Test Method for shake Extraction of Solid Waste with Water (ASTM D 3987-85). The test is essentially equivalent to the US EPA's Toxicity or TCLP tests except that the leaching solution consists of distilled water instead of an acetic acid buffer. the ASTM leach test was chosen because it consists of distilled water instead of an acetic acid buffer. The ASTM leach test was chosen because it is similar to the widely used regulatory tests, yet uses a leaching medium that is more representative of the type of leachate these wastes are likely to encounter than the leaching solution used in the EP Toxicity or TCLP tests.

Briefly, the test consists of mixing a given amount of solid with 20 times its weight in distilled water, shaking for 18 hours, and then analyzing the filtered leachate for the constituents of concern.

3.2 Comparison with other leach tests.

There are several standard batch leach tests that are in common use for evaluating leaching from wastes. The ASTM water leach test is compared with the EP Toxicity, TCLP and several other commonly used tests in Table 1. The tests are compared with the conditions that could have a major impact on the results of the test, and so are important for selecting which test to use.

3.2.1 Leaching Solution

The overriding influence on test results is the leaching solution used in the test. All these leaching tests are implicitly modeling what occurs to the waste after it is disposed. The choice of the leaching solution used is based on the disposal scenario being modeling. pH is generally considered a major controlling influence on the leaching of many heavy metals, notably those metals that occur in the primary drinking water regulations, and for that reason is a major parameter modeled in the leaching tests. The acidic solution used in the EP Toxicity and TCLP tests is designed to stimulate the acidic environment that occur in an actively decomposing municipal landfill during a short portion of its decomposition cycle. The water leaching tests are designed to stimulate what occurs in a disposal environment in which the waste itself controls the pH of the leaching solution, as would occur in a foundry and steel waste only landfill. High volume foundry and steel wastes are frequently disposed in mono landfills. In such disposal situations there is little if any putrescible material to generate the acids that lower the pH as in a municipal landfill. Rather the landfill leachate reflects the pH generated by the waste itself. Therefore, use of a water leaching test is a more realistic model of the pH controlling factors in the landfill than is the EP Toxicity or TCLP tests. Therefore, we recommend that a water leaching test be used.

3.2.2 Other Leaching Test Conditions.

As discussed above, in the interest of time and simplicity, the tests use some standard conditions that may not be very realistic but that are necessary for running the tests within a reasonable time frame. Conditions that can have a major influence on test results for some parameters are the solid to liquid ratio and the aeration, or oxidation-reduction potential, of the leaching solution. All the standard tests described in Table 1, with the exception of the AFS leaching tests, use similar solid to liquid ratios, and occur under aerated conditions.

3.3 Comparison of Leaching Test Results with Actual Field Leachate Data

There is relatively little data comparing leaching test results with actual field leaching concentrations, undoubtedly because it requires a relatively difficult and lengthy study for a good comparison to be made One study that has been done was a comparison of leaching test results with landfill lysimeter and groundwater samples around ferrous foundry landfills (Ham et al, 1986). this study has been mentioned in some of the previous testimony before the board.

The study found that leaching tests, both the EP Toxicity test and an EP Water test, overestimated the concentrations of the primary drinking water metals in the foundry waste leachate, but tended to underestimate the concentrations of very soluble or redox sensitive parameters (i.e., iron and manganese) in the foundry waste leachate and in the groundwater within the landfill. The concentration of dissolved iron or manganese in water is influenced by the oxidation state of the metal. Ferrous iron, the reduced state of iron, is generally much more soluble under naturally occurring conditions than is ferric iron, the more oxidized state. Thus iron is much more soluble under reducing conditions that it is under oxidizing conditions. Many landfills, foundry waste landfills included, have chemically reduced conditions within the landfill, and would tend to solubilize iron. However, most leaching tests are conducted under oxidizing conditions since they are shaken in air, and thus are not accurate models of the conditions occurring within the landfill. It is understandable, therefore, that the leaching tests are not good indicators of leaching of redox-sensitive parameters in a landfill.

4. PROPOSED CLASSIFICATION STANDARDS

In the proposed regulations, the results of the ASTM water leach test are compared with primary and secondary drinking water criteria for the inorganic parameters. Foundry and steel wastes are predominantly inorganic, and few organic parameters of concern are likely to be found in the wastes. Wastes are classified into one of four categories, based on the concentrations found in the leaching test: beneficially usable wastes, potentially usable wastes, low risk wastes, and chemical wastes. The criteria for classifying the wastes are based both on the federal drinking water standards and on the proposed State of Illinois groundwater standards.

The primary standards for beneficially and potentially usable wastes are the federal primary drinking water standards, or MCLs, for inorganic compounds (40 CFR 141.11) with mercury and silver not included in the list. Mercury and silver are not normally found in appreciable concentrations in steel and foundry wastes. For the low risk wastes, the concentrations are increased by a factor of five. Note that the hazardous waste criteria for Toxicity Characteristics wastes are also based on the MCLs, with the MCL criteria multiplied by a factor of one hundred. The proposed criteria are intended to be conservative, particularly since the Ham study found that leaching tests lend to overestimate the release of primary drinking water parameters from foundry wastes.

For the secondary standards, the criteria are derived either from the federal or state secondary drinking water criteria (SMCLs), or from the proposed groundwater standards (35 ILL Adm.Code 620.310). The basis for each of the proposed secondary parameter criteria is given in Table 2. The varying factors for choosing the criteria for the different parameters and levels is based on an estimation of the accuracy of the test for predicting concentrations and on the likelihood that these constituents may be in foundry and steel waste leachates. Chloride and sulfate are frequently present in waste as soluble parameters, and the water leach test may underestimate their initial concentrations in a waste leachate. Thus the factor of 2 was used for increasing the criteria between the potentially usable and low risk wastes. Iron and manganese commonly occur in leachates from foundry and steel wastes, and also in shallow groundwater systems. The criteria chosen for iron and manganese were based on the need to be not overly restrictive on the reuse of foundry and steel wastes. Iron and manganese are very commonly found in shallow groundwater, and are of concern for aesthetic reasons rather than for reasons related to human health. Thus having classification criteria for iron and manganese that are above the secondary drinking water criteria is not likely to have a deleterious environmental impact related to reuse or disposal of the foundry and steel wastes.

5. **SUMMARY**

The proposed method for classifying nonhazardous foundry and steel wastes uses a standard leaching test with criteria based on federal or state drinking water criteria. The test should provide a useful means of screening and classifying foundry and steel wastes under a standard set of leaching conditions.

. **REFERENCES**

Ham, R.K., W.C. Boyle, F.J. Blaha, D. Oman, D. Trainer, T.P. Kunes, D.G. Nichols, and R.R. Stanforth, 1986. Leachate and Groundwater Quality in and Around Ferrous Foundry Landfills and Comparison to Leach Test Results. Transactions Amer. Foundrymen's Soc. 94 935-941.