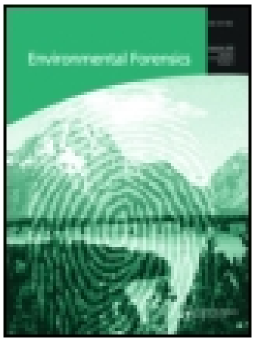

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Allocating remedial costs at a superfund site using risk magnitude, geographic information systems, and Monte Carlo analysis

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ABSTRACT
A method for allocating remediation costs among a number of potentially responsible parties (PRPs) was developed as part of a highly technical, complex Superfund litigation case involving a former hazardous waste disposal site located in the southern United States. The method was devised in response to questions from the U.S. district court regarding the volume and composition of wastes disposed, the “remedy drivers” (defined by the court as wastes or chemicals that most influenced the selection of the remedy), and the relationship between these factors and the various PRPs. A key element of the method was calculation of the *Risk Magnitude*, which was defined as the logarithm of the maximum concentration of a chemical in groundwater divided by its cleanup standard. Risk magnitude was linked to site remediation costs based on an analysis of remediation difficulty, data from remediation performance studies, and other sources. Key components of the allocation method included: (i) identifying remedy driver chemicals (RDCs) based on risk magnitude; (ii) quantifying the spatially adjusted cumulative risk magnitude associated with each RDC using a Geographic Information System (GIS); (iii) adjusting the risk magnitude to account for remediation difficulty of each RDC; (iv) correlating RDCs with the volume of different waste streams disposed by each PRP at the site; (v) developing an estimate for the waste volume for each PRP using multiple lines of evidence inside a Monte Carlo analysis; and (vi) apportioning final cleanup costs between PRPs based on their attributable volume and cumulative Risk Magnitude. The basic methodology was applied by the district court to develop a scientifically sound opinion that allocated site remediation costs between the PRPs—a decision that was upheld by the U.S. court of appeals. Though developed for a particular site, the method is adaptable, and its fundamental components could be applied to other sites where allocating remedial costs of complex chemical mixtures in environmental media is the objective.

KEYWORDS
Superfund; cost allocation; Monte Carlo; risk magnitude; groundwater

Introduction

Cost allocation under CERCLA

The Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), commonly referred to as Superfund, was enacted in 1980 to provide funds to clean up “uncontrolled or abandoned hazardous waste sites as well as accidents, spills, and other emergency releases of pollutants and contaminants into the environment” (US Environmental Protection Agency, 2017). The United States Environmental Protection Agency (USEPA) has the authority to identify, and hold financially accountable, potentially responsible parties (PRPs). These parties can then seek contributions for costs incurred from other PRPs, which is typically referred to as a Section 113(f) claim.

Allocation factors

Courts apportioning CERCLA costs typically do so by examining six general factors, created by former Senator Al Gore (Graves et al., 2000). The “Gore Factors” are:

1. the ability to distinguish a party’s contribution to the discharge, release, or disposal of hazardous waste;
2. the volume of hazardous waste disposed;
3. the toxicity of the hazardous waste disposed;
4. the degree of involvement of a party in the generation, treatment, storage, and disposal of hazardous waste;
5. the waste management practices of the party; and
6. the party’s degree of cooperation with regulatory agencies.

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However, these factors are not exhaustive, and courts can consider any equitable factor they deem appropriate, particularly in situations where the usefulness of the Gore factors may be limited. CERCLA allocation can be complex, and there is no universal method for determining waste ownership amongst PRPs. Some previously used strategies are briefly explained below.

Allocation methods described in the literature

While numerous cost allocation methods have been proposed and/or applied based on site-specific conditions, this section describes several common approaches as described in the literature. Some key considerations in the selection of an allocation method are: the mass or volume of waste contributed by each party, the relative toxicities of constituents of concern (COCs) present in the waste, the separability of the waste by party and geographic area, the effectiveness of remediation at reducing toxicity or volume of the waste, and the quality of information available to the allocator for assessing each of these factors.

The simplest approaches utilize mass-only and/or volume-only allocation. In these approaches, the allocation simply considers a party's percentage of total waste mass or volume contributed by all the PRPs (e.g., Graves et al., 2000). While this approach is advantageous for ease of application, it fails to consider COC-specific factors such as relative toxicity and remediation effectiveness for the individual COCs and therefore would be most applicable to allocation of single constituent wastes. The approach can be furthered by combining mass and volume into a single metric, which may be a more accurate allocation method in some cases. For instance, a concentrated (i.e., high mass) but compact (i.e., low volume) groundwater plume will be inexpensive to pump but expensive to treat, and vice versa (Graves et al., 2000).

With a "stand-alone" cost allocation method, as described by Marryott et al. (2000), each PRP's cost allocation is proportional to the cost they would have incurred had they engaged in their own individual remediation project. If the PRPs collectively implement a single remediation system, they benefit from the economies of scale and the elimination of the redundancy of multiple systems, thus producing lower costs compared to the sum of individual remediation projects. Therefore, to determine the final cost attributable to each PRP, the total cost of the collective remediation system is allocated by the proportional cost share that each PRP would have incurred if they had implemented their own separate (i.e., stand-alone) remedial system (Marryott et al., 2000).

Finally, a risk-based allocation approach evaluates the potential harm to human health and the environment by a contaminant release. Murphy (1996) and Mink et al. (1997) describe the approach as being based on the need for a remedy rather than the cost. The driving factors for risk, therefore, are the contaminant type and toxicity, as well as the concentration of contaminants that need to be treated and/or removed. These risk-based approaches may be desirable when the range of risk between COCs at a location are vastly different from each other.

Comparison of allocation methods

When allocations are based solely on either volume or mass, not only is the other factor ignored, but relative toxicity is also not explicitly considered. For example, the PRP may be given an unfair allocation cost when evaluating mass alone, due to a small, high-concentration plume. Conversely, a volume-only allocation may penalize PRPs with low concentration, dispersed plumes. While allocation based on volume and mass together offers one approach to resolving potentially competing mass and volume data, there are still issues associated with the method, such as the potential risk of each PRP's contamination to human health and the environment if relative toxicities of COCs are substantially different (Graves et al., 2000). Stand-alone cost allocation works well when there is a clear documentation of waste disposed at the site, the waste is similar in toxicity, a similar level of remediation is needed, and the contamination is present in the same area. If waste disposal and associated risk data are not available or incomplete, then other methods may be more appropriate.

This article describes a large, multiple party, complex Superfund Site where a site-specific allocation methodology was developed by the authors of this manuscript for consideration by the U.S. district court. The allocation methodology that was developed included incorporation of several elements that, to our knowledge, had not previously been used within the framework of cost allocation. The primary elements in this regard included risk magnitude to account for COC-specific risk and remediation cost, Monte Carlo analysis to account for uncertainty in disposal volumes, and Geographic Information Systems (GIS) to account for spatial analyses of the waste and remediation effectiveness. Whereas a previous study (Dankwah, 2010) described the use of probabilistic models to determine remedial soil volumes, the method described herein incorporated additional variables to make it more robust given the site-specific complexities. The following sections present brief overviews of the site and the litigation matter as they relate to cost allocation.

Site history

The Superfund site was located on a 500+ acre plot of land that was used as an illicit waste disposal site between the late 1960s and early 1970s. Two waste hauling companies that transported waste from the PRPs' facilities would utilize the illicit site when nearby authorized disposal sites were not accepting waste. The waste haulers dumped chemical waste into six unlined pits that were located along a 2.5-mile road that traversed the site. These operations impacted the road, surrounding soil, and groundwater with chemicals. Illicit disposal activities ceased at the site in approximately 1974.

Environmental investigations began at the site in 1986. Remedial action started in January 1988 when soil was excavated to depths of up to 5 feet below ground surface along 1,800 feet of the road that transects the site. The excavated area was then backfilled with soil and paved. Remedial actions to protect and restore groundwater at the site began in June 1997 and consisted primarily of treatment by in situ technologies. The primary remedial technologies used were aerobic in-situ bioremediation and soil vapor extraction. The in situ bioremediation process consisted of groundwater extraction, enrichment of the extracted water with nutrients and oxygen, and re-injection to contaminated zones. Other remediation technologies that were implemented at selected "hot spots" included in situ thermal desorption, in-situ chemical oxidation, and bioaugmentation. In addition, hundreds of groundwater monitoring wells were installed and tested at the site. Active remediation ended in the mid-2000s with a demonstration of technical impracticability for further contaminant removal in groundwater, and the site transitioned to management using monitored natural attenuation and institutional controls to limit groundwater plume migration and exposure to potential receptors. Thus, the allocation method described herein was applied primarily to incurred costs for groundwater protection and restoration.

Litigation overview

In the early 2000s, two parties that had been identified by the USEPA as being liable for the remediation costs at the site initiated cost recovery litigation against numerous other PRPs. The complex nature of the disposal history meant that the simple allocation methods described above were inadequate for allocation. The historical site documents were sometimes vague and/or fragmentary, making any allocation based on volume or mass alone infeasible. Waste haulers disposed of waste from numerous different chemical plants into one or more unlined pits situated at the site, resulting in mixing of waste from the different PRPs. Stand-alone cost allocation, therefore, would be difficult to enforce amongst the PRPs because the different waste streams had different cleanup standards, which had a large impact on ultimate remediation costs. For these reasons, and as described further below, development of a site-specific allocation approach was needed in this case to allow incorporation of numerous factors into the final allocation framework.

In 2006, due to the complexity of the site and numerous allocation approaches offered by the PRPs, the federal judge presiding over the case appointed a technical expert on behalf of the court to determine an equitable allocation of remediation costs between the PRPs. This technical expert was charged with developing a fair and equitable allocation of site remediation costs based on the available historical documentation and known site history.

Key questions presented by the court to the technical expert were:

- *What was the total volume of waste disposed at the site, and how much was disposed by each of the PRPs?*
- *What percent of the waste remained after site remediation?*
- *What was the chemical composition of the waste disposed at the site?*
- *Is the site divisible geographically or by chemical type?*
- *What were the remedy drivers for the cleanup activities at the site?*
- *What was the relationship between each PRP's waste and the cost of the remedies?*

Allocation method

To answer these questions, the technical expert combined three elements (i.e., Risk Magnitude analyses, GIS analyses, and Monte Carlo analyses), as shown in

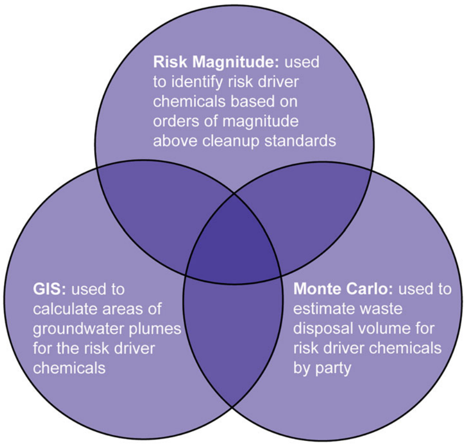


Figure 1. Allocation approach integrating risk magnitude, Geographic Information System (GIS), and Monte Carlo Analysis.

Figure 1, to develop a site-specific allocation approach consisting of the following steps:

1. Calculate the risk magnitude for each sampled constituent at each monitoring well;
2. Identify remedy driver chemicals (RDCs) based on Risk Magnitude;
3. Perform spatial analysis of risk magnitude associated with each RDC in each waste area using a GIS to quantify the area-weighted cumulative risk magnitude;
4. Adjust the area-weighted cumulative risk magnitude to account for the remediation difficulty of each RDC;
5. Correlate the RDCs with the different waste streams disposed at the site;
6. Develop an estimate of the waste volume for each PRP using multiple lines of evidence within a Monte Carlo analysis; and
7. To arrive at a final allocation, apportion the adjusted cumulative risk magnitudes calculated for each chemical to each PRP based on the PRP's volume contribution of the chemical, then sum the apportionment of all chemicals contributed by each PRP.

Each step is detailed in the following sections.

Quantify "risk magnitude"

A concept of risk magnitude was developed to identify the key contaminants driving the need for remediation at the site (i.e., the RDCs). Risk Magnitude was computed for each contaminant that was tested in groundwater at the site according to the equation below:

$$\text{Risk Magnitude} = \log_{10}\left(\frac{\text{Maximum historical concentration}}{\text{Regulatory cleanup standard}}\right)$$

Maximum concentrations for each contaminant at each monitoring well were divided by the regulatory cleanup standard for each constituent that was defined in the site's Record of Decision issued by the USEPA, resulting in a standard-normalized concentration value. The common log of the standard-normalized concentration value yielded the risk magnitude. Maximum historical concentrations were utilized so that the analysis would reflect pre-remediation conditions (i.e., the potential cost liability before remedial action occurred). This calculation established the number of Orders of Magnitude (OoM) concentration reduction required to reach the applicable cleanup standard and is a cornerstone of the overall allocation method because remediation costs can be related to the number of OoMs, as discussed below.

Relationship between risk magnitude and remediation cost

Risk magnitude (the OoMs above the cleanup standard) has a practical significance with regard to remediation costs. First, groundwater datasets are often better represented by a log-normal concentration distribution rather than a normal (Gaussian) distribution (Helsel and Hirsch, 2002). This is due to the extremely wide variation in groundwater concentrations at most sites, particularly at sites such as the one in question where large amounts of waste were disposed. The log transformation normalizes the data in a way that makes for simple comparison of the potential harm between contaminants (Newell et al., 2011).

Second, several studies (e.g., McGuire et al., 2006; McGuire et al., 2016) have observed that in-situ remediation projects generally reduce groundwater concentrations in the source zone by approximately one OoM (i.e., reduce concentrations by a factor of 10, which is equivalent to a 90% reduction). This "rule-of-thumb" arose from a study of numerous chlorinated solvent sites where groundwater remediation had been attempted using in situ remediation technologies (McGuire et al., 2006). McGuire et al. (2006) found that the median percent reduction in groundwater concentrations at 59 sites was 96% for in situ biodegradation; 88% for chemical oxidation; 98% for thermal treatment; and 96% for surfactant/cosolvent treatment. A subsequent study of 235 chlorinated solvent sites confirmed this finding and observed that the mean reduction in concentration achieved by in situ remediation technologies was 91%, or about 1.1 OoM (McGuire et al., 2016).

Application of a second remediation treatment or alternative technology could, in theory, decrease the concentrations by another 90% (resulting in a total concentration reduction of 2 OoMs), but the cost of this second technology would be approximately equal to the cost of the first technology. This assumption that each subsequent technology applied would achieve an approximate 1 OoM reduction in groundwater concentrations is supported by a study of 14 sites with treatment trains, where two technologies were implemented in succession and the combination of technologies resulted in a median OoM reduction of 2.3 OoM, whereas the single technology sites achieve a median OoM reduction of 1.1 (McGuire et al., 2016). Therefore, the cost allocation model presented here relates remediation costs directly to risk magnitude (i.e., the number of OoMs above the cleanup standard). It should be noted that this relationship is most applicable to remediation using treatment technologies to reduce COC concentrations and may not apply to certain containment approaches designed to physically isolate waste in bulk (e.g., excavation and landfilling, slurry walls).

As an example of the risk magnitude/OoM concept, consider hypothetical Site A with a monitoring well containing benzene at a concentration of 50 micrograms per liter (μg/L). Hypothetical Site B has a monitoring well with a benzene concentration of 500,000 μg/L. If the regulatory cleanup standard for benzene at both sites is 5 μg/L, Site A exceeds the benzene standard by 10 times, while Site B exceeds the benzene standard by 100,000 times. Using the formula presented above, Site A has a risk magnitude of 1 for benzene (50 μg/L vs. 5 μg/L; or 1 OoM over the standard), while Site B has a risk magnitude of 5 for benzene (500,000 μg/L vs. 5 μg/L; or 5 OoM over the standard). The risk magnitude/OoM cost model presented here indicates that Site B would be about 5 times more costly to remediate due to the higher concentration exceedance above the groundwater cleanup standard, assuming the sites are the same size and all other factors being equal.

Remedy driver chemicals (RDC) identification

In this particular matter, which involved a site where numerous chemicals had been detected in groundwater samples, the federal court specified identification of the remedy drivers, defined by the court as "those wastes, or chemical constituents in the wastes, that most influence the selection of the remedy, measurement of success or failure of the remedy, and the costs of the remedy." On this basis, contaminants that did not present substantial risk, and therefore did not drive cleanup costs, were eliminated from further consideration in the analysis. The following two criteria were utilized to eliminate COCs from consideration as RDCs:

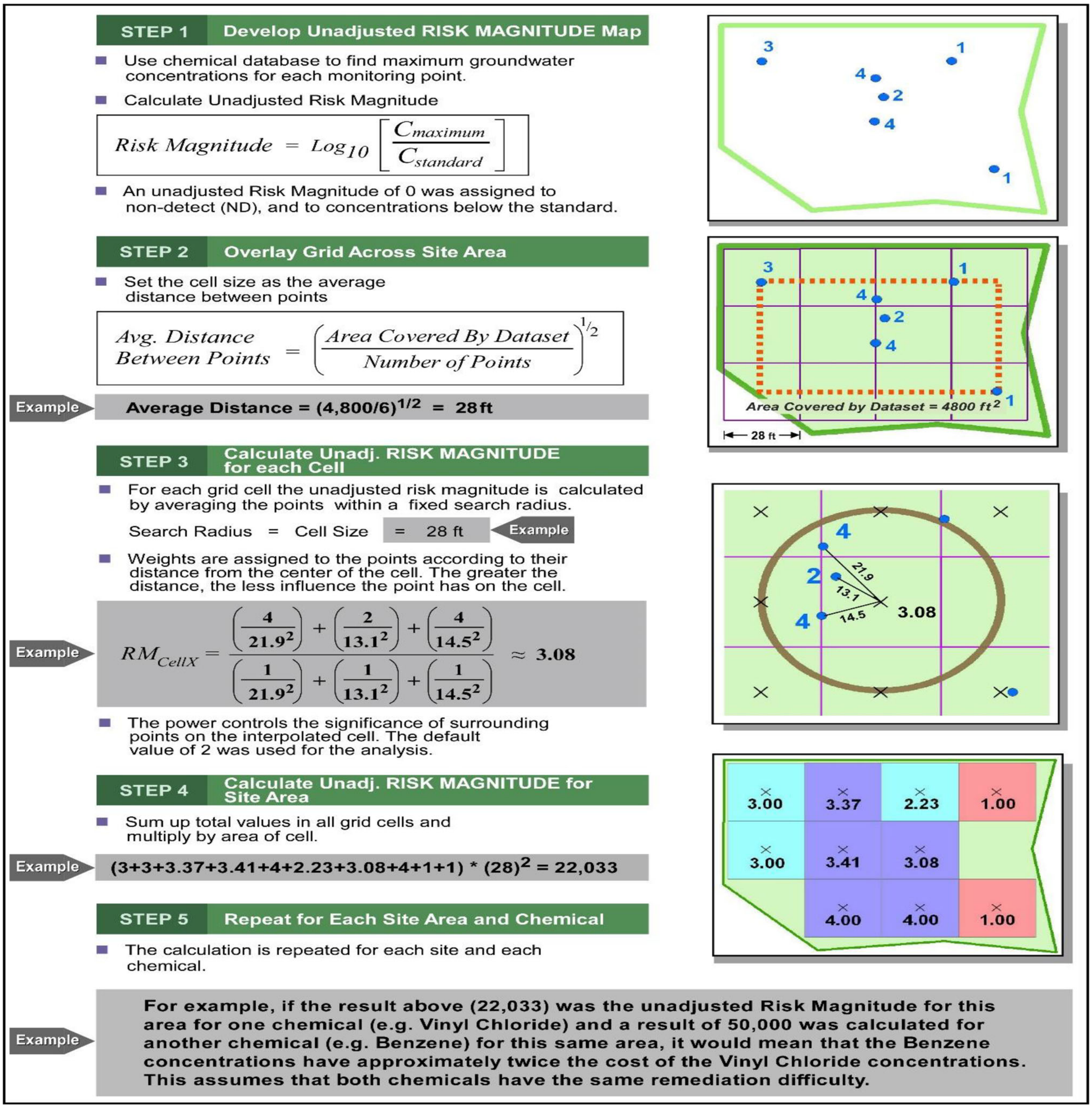
1. The median risk magnitude (the middle risk magnitude value from all the wells at the site) was less than zero (i.e., below the cleanup standard), or
2. The chemical was detected in fewer than 20% of the groundwater samples.

This approach eliminated chemicals with low risk or chemicals that were only found at the site in limited quantities. While it is possible for chemicals with low risk or low frequency of detection to drive some portion of cleanup costs at a particular subarea, the distribution of RDCs having high risk and high frequency of detection was widespread at the subject site, indicating that the vast majority of site risk was captured by the retained chemicals. At other sites, site-specific information could be incorporated at this stage to determine whether specific low risk or low frequency chemicals should be retained for particular subareas. Alternatively, at sites with few chemicals, this screening step could be eliminated from the allocation approach.

Spatial analysis and cumulative risk magnitude using GIS

While treatment-oriented groundwater remediation costs are typically directly related to the risk magnitude, the amount of affected media requiring remediation is also a key cost driver. To account for spatial distribution of risk magnitude in the allocation, a GIS was used to calculate the areas of the groundwater plumes and the area-weighted risk magnitude for each of the retained RDCs. A GIS is a special-purpose digital database in which spatial coordinates are the primary means of reference. Since data are related geographically, data can be analyzed spatially (such as the calculation of various areas associated with the data). One of the key technical advantages of a GIS system was exploited at the site to calculate not only the areas of the groundwater plumes, but also the areas associated with different Risk Magnitudes for each retained chemical.

Figure 2 presents the overall method used to spatially analyze and cumulate total Risk Magnitudes for



STEP 1 **Develop Unadjusted RISK MAGNITUDE Map**

- Use chemical database to find maximum groundwater concentrations for each monitoring point.
- Calculate Unadjusted Risk Magnitude

$$Risk\ Magnitude = Log_{10}\left[\frac{C_{maximum}}{C_{standard}}\right]$$

- An unadjusted Risk Magnitude of 0 was assigned to non-detect (ND), and to concentrations below the standard.

STEP 2 **Overlay Grid Across Site Area**

- Set the cell size as the average distance between points

$$Avg.\ Distance\ Between\ Points = \left(\frac{Area\ Covered\ By\ Dataset}{Number\ of\ Points}\right)^{1/2}$$

Example Average Distance = $(4{,}800/6)^{1/2}$ = 28 ft

STEP 3 **Calculate Unadj. RISK MAGNITUDE for each Cell**

- For each grid cell the unadjusted risk magnitude is calculated by averaging the points within a fixed search radius.

Search Radius = Cell Size = 28 ft **Example**

- Weights are assigned to the points according to their distance from the center of the cell. The greater the distance, the less influence the point has on the cell.

Example

$$RM_{CellX} = \frac{\left(\frac{4}{21.9^2}\right) + \left(\frac{2}{13.1^2}\right) + \left(\frac{4}{14.5^2}\right)}{\left(\frac{1}{21.9^2}\right) + \left(\frac{1}{13.1^2}\right) + \left(\frac{1}{14.5^2}\right)} \approx 3.08$$

- The power controls the significance of surrounding points on the interpolated cell. The default value of 2 was used for the analysis.

STEP 4 **Calculate Unadj. RISK MAGNITUDE for Site Area**

- Sum up total values in all grid cells and multiply by area of cell.

Example $(3+3+3.37+3.41+4+2.23+3.08+4+1+1) * (28)^2 = 22{,}033$

STEP 5 **Repeat for Each Site Area and Chemical**

- The calculation is repeated for each site and each chemical.

Example **For example, if the result above (22,033) was the unadjusted Risk Magnitude for this area for one chemical (e.g. Vinyl Chloride) and a result of 50,000 was calculated for another chemical (e.g. Benzene) for this same area, it would mean that the Benzene concentrations have approximately twice the cost of the Vinyl Chloride concentrations. This assumes that both chemicals have the same remediation difficulty.**

Figure 2. Method for calculating the cumulative risk magnitude (unadjusted for remediation difficulty) for an RDC within a site area.

each RDC and site area. Using a database provided in the trial evidence, the groundwater data were first analyzed to identify the risk magnitude by RDC at each groundwater monitoring well. Next, each of the six identified waste disposal areas at the site were overlain with a grid system. The size of the grid was related to the density of the monitoring wells installed in each waste area and generally, the six waste disposal areas had grid cell sizes that ranged from 46 feet to 108 feet. Next, an inverse-distance weighted calculation method was used to calculate the risk magnitude within each grid cell. This method averages the groundwater data within a fixed radius distance from the center of each cell, with increased weight placed on monitoring points located closer to the cell center. Once the risk magnitude had been estimated at each grid cell, the cumulative risk magnitude for each RDC at each waste area was calculated by summing the grid cell values and multiplying by the area of the grid cell.

Remediation difficulty adjustment

The following remediation technologies were applied at the site in an attempt to achieve the site cleanup goals for groundwater protection: groundwater extraction and treatment, in situ bioremediation, soil vapor extraction, in situ thermal desorption, and in situ chemical oxidation. However, despite these efforts, the USEPA concluded that, "restoration of the impacted ground water and overlying soils has been determined to be technically impracticable."

These remediation efforts, however, allowed comparison of the difficulty, or recalcitrance, in remediation for different chemicals by comparing groundwater concentrations before and after remediation. Only monitoring wells with groundwater data before remediation (defined as starting in June 1997) and groundwater data after remediation (defined as after January 2004) were analyzed. A relative scale of "remediation difficulty" was then developed for each of the retained RDCs using a reduction in Risk Magnitude from before to after remediation as the metric. The remediation difficulty metric of each RDC was then normalized to that of vinyl chloride, as this chemical had the greatest response to remediation (i.e., lowest remediation difficulty). The normalization was accomplished by dividing each chemical's reduction in median Risk Magnitude from before to after remediation by the value for vinyl chloride and then taking the reciprocal (divide into one).

The area-weighted cumulative risk magnitude values calculated in the step above were then adjusted to reflect the normalized remediation difficulty for each RDC. For example, if the normalized remediation difficulty for a particular RDC was 1.6, the area-weighted cumulative risk magnitude values estimated by the GIS system were increased by a factor of 1.6.

Correlate the RDCs with the different waste streams

A weight of evidence approach was used to determine which of the RDCs were contained in wastes from each PRP's facilities. Actual waste composition data from the time period of waste disposal, along with Findings of Fact provided by the U.S. federal court, were used as the primary line of evidence to determine the presence or absence of a particular RDC in a particular PRP waste stream. If historical waste records showed that a particular RDC was present in a waste stream that was disposed at the site, then that chemical was assigned to the PRP. If no waste composition data were available, or if the available description or analyses were not chemical specific, then secondary lines of evidence were reviewed to determine RDC assignments based on the expert's professional judgment. These secondary lines of evidence included Toxic Release Inventory, RCRA hazardous waste, and soil/groundwater data from the PRP's facilities. A detailed description of each scenario is outside the scope of this paper.

Determining RDC disposal volumes using Monte Carlo analysis

Incomplete and sometimes fragmentary data were available to estimate the volumes of waste contributed by each PRP to the site. To manage uncertainty in the estimation process, available historical documents and testimony regarding disposal volumes from each of the PRP's facilities were analyzed and summarized. The quality and completeness of these records varied significantly between the PRPs, increasing the complexity of the volume analysis. For example, some PRPs had relatively complete waste records that provided volume estimates within a tight range while others were relatively meager, resulting in a relatively large range of potential disposal volumes. One of the key technical challenges for the volume analysis was to develop an estimation method that could reflect the different types of available data (e.g., tight-range, low-uncertainty vs. wide-range, high uncertainty). To explicitly account for the inherent uncertainty based on available data, the available disposal volume estimates for each PRP's facility were reduced to a statistical distribution rather than a single estimated value. These statistical distributions allowed use of a Monte Carlo analysis for estimating a final waste volume and uncertainty associated with each PRP.

Each PRP's facility waste volume data were analyzed and then described by either a uniform or triangular statistical distribution. These distributions reflected the available information about the range of the disposal volumes and the probability that specific volumes were actually disposed at the illicit disposal site. With a uniform distribution, the limits of the possible values are known (a minimum and a maximum), but there is no information on the most likely value within that range. The triangular distribution is structured so that a disposal volume estimate is forced to be between a minimum and maximum value, but with increasing chance of using a disposal volume near the mode (i.e., the most common value, which

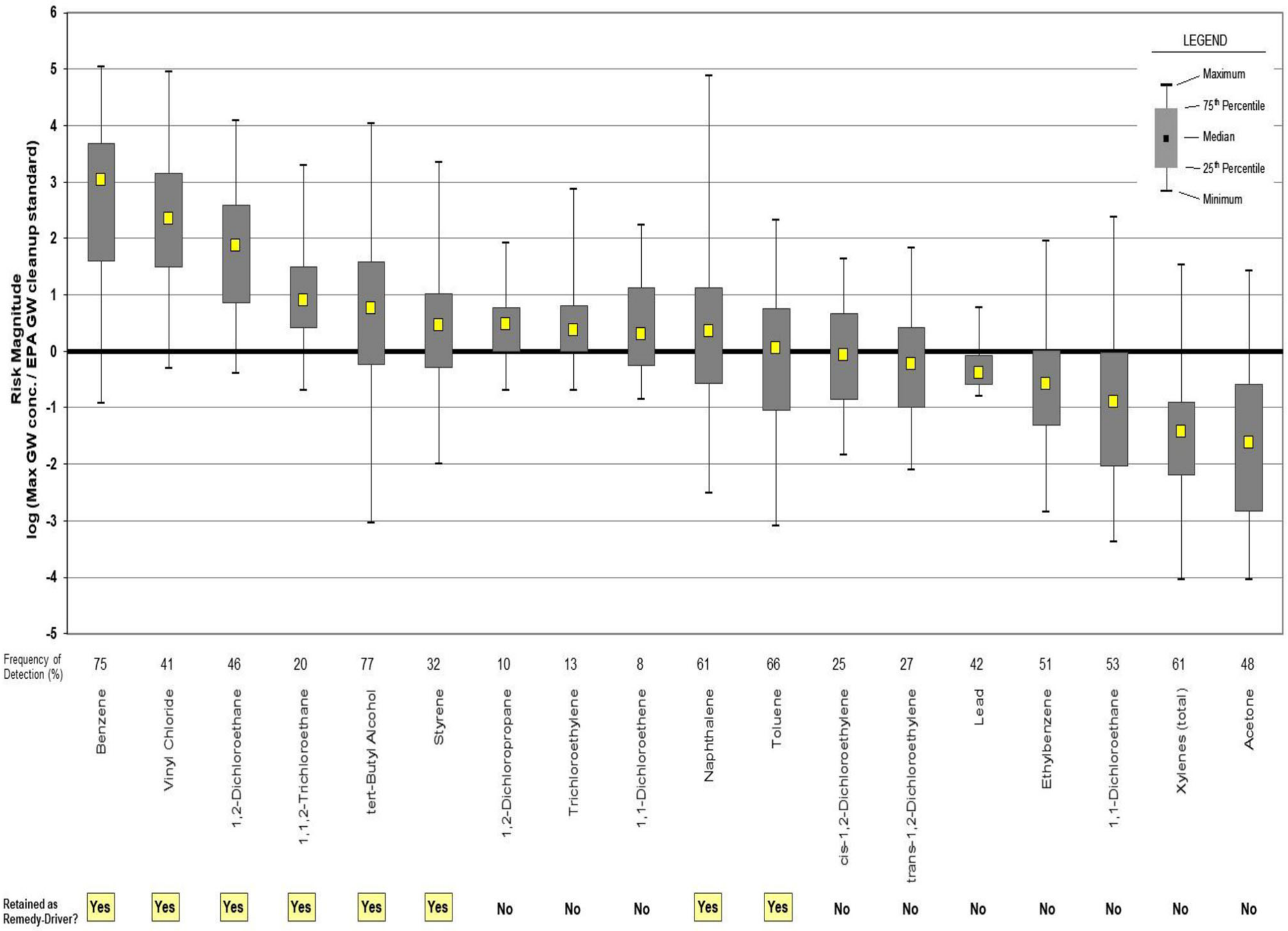


Figure 3. Determination of RDCs based on median risk magnitude and detection frequency.

does not necessarily have to be the average of the minimum and maximum). The triangular distribution accounts for the fact that some values are more likely than others, but that there is some chance an extreme value could occur.

The next step was to use the statistical distributions within a Monte Carlo analysis (USEPA, 1997) to estimate the disposal volumes and associated uncertainty from each facility. The Monte Carlo method was applied for total disposal volumes and for each RDC individually. Similar to the total volume estimates, statistical distributions (uniform or triangular) for the RDC volume disposed at the site were developed by matching the waste stream information with each RDC. The statistical distributions were then entered into a Monte Carlo software package add-in for Microsoft Excel (RiskAmp, 2019). The Monte Carlo analysis was performed using 500,000 realizations with the Latin Hypercube option selected, which is widely used in Monte Carlo simulation because it can drastically reduce the number of runs, and therefore computing resources, necessary to achieve a reasonably accurate result. The median volume produced from the Monte Carlo analysis was selected as the final disposal volume for each PRP and RDC, with the Monte Carlo approach providing information on the uncertainty in these volumes.

Final allocation using cumulative adjusted risk magnitude and RDC volume

Combining the results from quantifying cumulative risk magnitude for each RDC using GIS (and adjusted for remediation difficulty), identifying RDCs associated with each PRP based on available trial evidence and other documentation, and calculating volume of RDCs disposed at the site by each PRP based on Monte Carlo analysis, an overall allocation model was constructed by apportioning the cumulative Risk Magnitudes by PRP based on their waste composition and volume disposed at the site. This allocation represented the relative post-removal action and pre-remediation cost for managing the site.

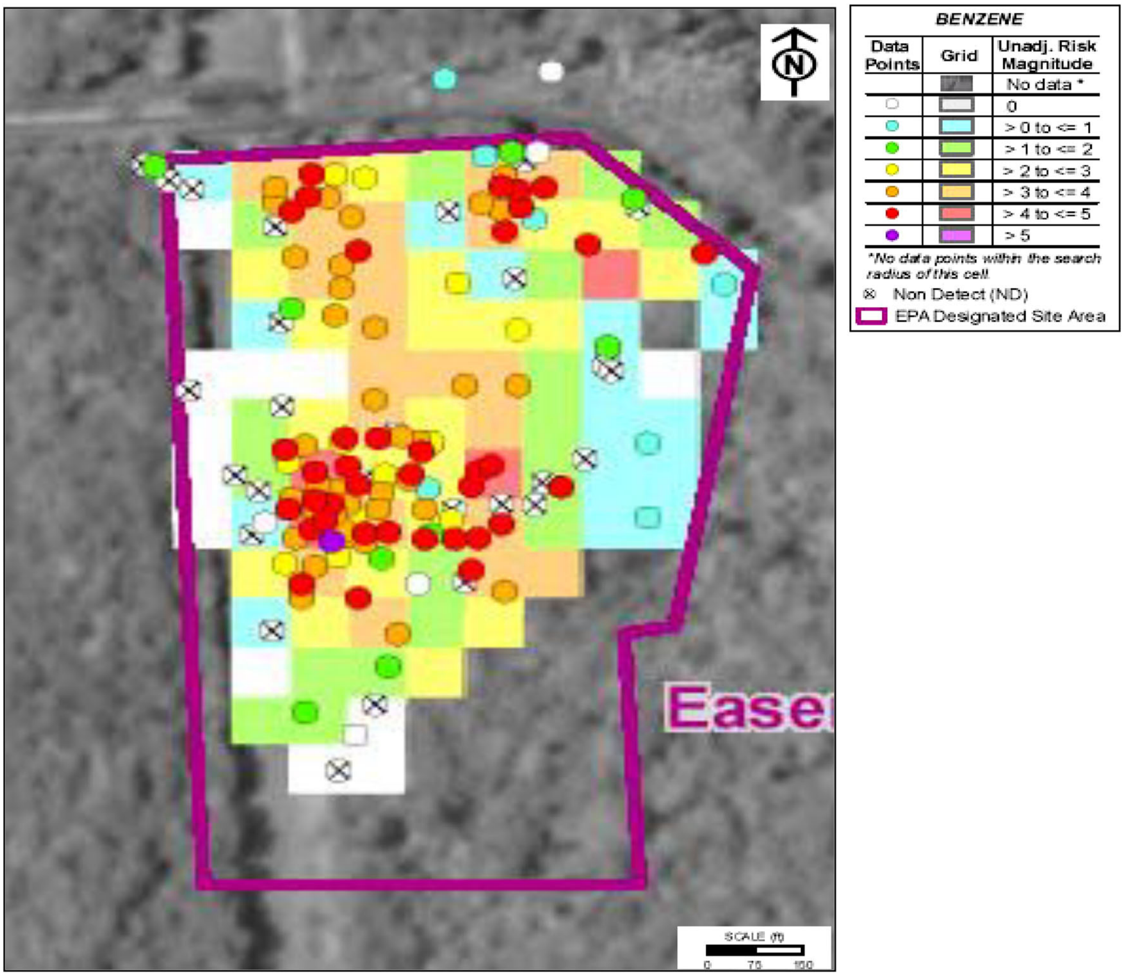


Figure 4. Risk magnitude map showing groundwater monitoring wells and geographic distribution of Risk Magnitude determined by inverse weighting calculation within the GIS.

Results and discussion

Identified RDCs

The box-and-whisker diagrams presented in Figure 3 summarize the risk magnitude distributions for the 18 chemicals of potential concern at the site for all monitoring wells; the frequency of detection is also shown for each chemical on Figure 3. Overall, benzene had the highest risk magnitude (maximum of 5.0 and median of 3.0) and acetone had the lowest risk magnitude (maximum of 1.4 and median of −1.6). The following eight constituents, which had a median risk magnitude exceeding zero (i.e., median of maximum concentrations exceeding the cleanup standard) and a detection frequency of 20% or greater, were retained as the RDCs for cost allocation: benzene; tert-butyl alcohol (TBA); vinyl chloride; styrene; 1,2-dichloroethane (1,2-DCA); naphthalene; 1,1,2-trichloroethane (1,1,2-TCA); and toluene.

Cumulative risk magnitude for each RDC

The process described in Figure 2 was implemented in the GIS to calculate cumulative risk magnitude for each of the eight RDCs at each of the six waste areas. Performing the calculation using GIS made the calculation easily repeatable for the eight retained groundwater chemicals at the six waste areas. As a representative example, Figure 4 illustrates the results of the calculation for benzene within each grid cell based on the inverse-distance weighting approach at one waste area.

Adjusting risk magnitude for remediation difficulty

The box-and-whisker diagrams presented in Figure 5 summarize the change in risk magnitudes observed at each monitoring well across the site for each RDC as a result of the remedial efforts that were implemented

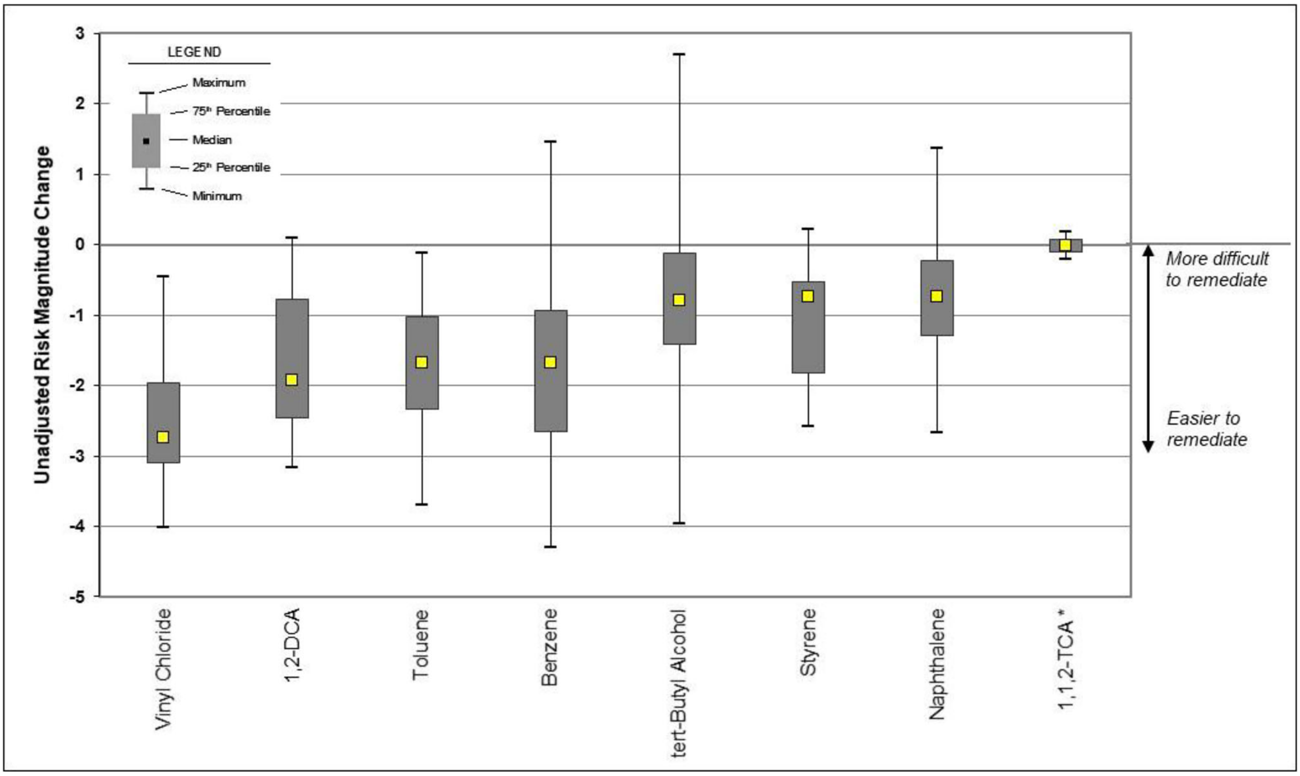


Figure 5. Change in risk magnitudes calculated for each RDC at each monitoring well from before remediation (June 1997) to after remediation (January 2004). Analysis excluded non-detects. Median values used to establish "remediation difficulty" multiplicative factor for each RDC.

between June 1997 and January 2004. Remedial actions had the largest effect on vinyl chloride, as seen by the largest reduction in median risk magnitude from before to after remediation of −2.75 OoM. A −2.75 OoM reduction of risk magnitude is equivalent to a contaminated groundwater concentration of 10,000 µg/L being reduced to 18 µg/L by remedial efforts. Despite intensive remediation efforts at the site, numerous groundwater monitoring wells still exhibited exceedances of the USEPA cleanup targets, even for chemicals that responded well to remediation, such as vinyl chloride.

An alternative method based on chemical-physical properties of each RDC was also evaluated. This properties-based method, using data from the scientific literature rather than site-specific data, yielded results consistent with the site-specific change in risk magnitude method. Because the change in risk magnitude approach was based on actual site data, it was used, with a single exception, in lieu of the chemical-physical properties methodology. The exception was for 1,1,2-TCA, which had insufficient data (only 2 monitoring wells with data before and after remediation) to calculate a change in risk magnitude. Thus for 1,1,2-TCA, the chemical-physical properties data were used to estimate the remediation difficulty.

Because vinyl chloride showed the largest reduction in median risk magnitude due to remedial actions, the generalized remediation difficulty for each RDC was normalized to vinyl chloride. The resulting normalized remediation difficulty values were used as multiplicative factors to "adjust" the cumulative risk magnitudes for the remediation difficulty of each RDC. For example, a remediation difficulty factor of 2 means that remediation is effectively twice as difficult (costly) as a value of 1. The remediation difficulty adjustment factors ranged from 1.4 for 1,2-DCA to 3.8 for naphthalene. Table 1 summarizes the change in risk magnitude due to remediation for each RDC, remediation difficulty multiplicative factors, cumulative risk magnitudes adjusted for remediation difficulty, and final allocation apportionment percentages by RDC based on the cumulative adjusted risk magnitude.

Volume of RDCs for each PRP

Monte Carlo analysis was applied for each RDC and for each PRP. Mining disposal data from the historical waste stream information, including both volumes and composition, were used, when available, to create individual distributions for the range of possible disposal volumes. Results summarized on Table 2

Table 1. Summary of change in risk magnitude due to remediation, remediation difficulty multiplicative factors, cumulative risk magnitudes adjusted for remediation difficulty, and final allocation apportionment percentages by RDC based on cumulative adjusted risk magnitude.

Remedy driver chemical	Reduction in median risk magnitude	Remediation difficulty factor	Cumulative adjusted risk magnitude	Allocation distribution by RDC (%)
Benzene	−1.68	1.6	3,195,651	28.9
TBA	−0.80	3.4	3,050,426	27.5
1,2-DCA	−1.93	1.4	1,137,176	10.3
Vinyl chloride	−2.75	1.0	1,087,465	9.8
Naphthalene	−0.73	3.8	1,047,660	9.5
1,1,2-TCA	*	3.8	640,667	5.8
Styrene	−0.74	3.7	621,982	5.6
Toluene	−1.69	1.6	289,371	2.6

*Insufficient data were available to calculate a reduction in risk magnitude for 1,1,2-TCA; therefore, a remediation difficulty factor of 3.8 was assigned based on chemical-physical properties.

Table 2. Volume (gallons) of each RDC attributable to each PRP based on waste composition and disposal information evaluated through Monte Carlo analysis.

PRP #	Benzene	TBA	1,2-DCA	Naphthalene	Vinyl chloride	1,1,2-TCA	Toluene	Styrene
PRP #1	86,771	1,134,088	–	–	–	–	86,771	95,070
PRP #2	587,529	–	–	587,530	–	–	587,529	–
PRP #3	509,700	–	–	509,700	–	–	527,411	–
PRP #4	–	–	62,294	–	62,294	62,294	–	–
PRP #5	21,006	–	54,500	21,006	54,500	54,500	21,006	–
PRP #6	40,000	–	–	40,000	–	–	40,000	–
PRP #7	–	–	–	24,875	–	–	–	–
PRP #8	–	–	–	–	–	–	81,077	–
Total	1,245,006	1,134,088	116,794	1,183,112	116,794	116,794	1,343,794	95,070

"–" indicates the RDC was not in the waste stream sourced from the PRP.

Table 3. Final cost allocation to each PRP after apportioning RDC risk magnitude and volume.

PRP #	Cost allocation (%)
PRP #1	35.28
PRP #2	19.49
PRP #3	16.94
PRP #4	13.81
PRP #5	12.78
PRP #6	1.33
PRP #7	0.20
PRP #8	0.16

demonstrate that toluene, benzene, naphthalene, and TBA were disposed at the site in similar volumes (ranging from approximately 1.13 to 1.34 million gallons). The chlorinated compounds and styrene are disposed at volumes approximately an order of magnitude less (ranging from approximately 0.095 to 0.12 million gallons). From these data, the percentage RDC volume contributed by each PRP was calculated.

Final cost allocation

When the results from the RDC analysis, risk magnitude calculations, and waste disposal volumes were combined, an overall cost allocation was constructed by apportioning the cumulative adjusted risk magnitude by RDC to each PRP. The apportioned share for each RDC was then summed to arrive at the final cost allocation for each PRP. The results are summarized on Table 3. Note that four significant digits were used in the final allocations to provide a result that totals exactly to 100%. The precision of the calculation is much less than four significant figures.

The final allocation results indicate that PRP #1 had the largest share of the remediation cost. This high share is primarily because this PRP contributed 100% of the TBA, which had the second largest cumulative adjusted risk magnitude share of 27.5% (Table 1), meaning that PRP #1 was responsible for 100% of the 27.5% TBA share. Similarly, PRP#1 contributed 100% of the styrene, which had a 5.6% risk magnitude share. PRP #1 also contributed 7% of the total benzene disposed, which had the highest RDC risk magnitude of 28.9%, resulting in a 2.0% overall cost allocation (i.e., 7% times 28.9% = 2.0%). PRP #1 also contributed toluene (2.6% RDC share times 6.5% volume share = 0.2% total cost allocation). Summing the total cost allocation share for each RDC resulted in an overall cost allocation of 35.28% for PRP #1 (i.e., 27.5% + 5.6% + 2.0% + 0.2%).

On the other hand, PRP #6 contributed 3 of the 8 RDC chemicals, but at relatively low volumes, resulting in the third lowest overall cost allocation of 1.33%. PRPs #7 and #8 each contributed only 1 RDC with relatively low cumulative risk magnitude shares, and at relatively low volumes, resulting in very low final allocations of 0.20% and 0.16%, respectively.

As part of the Monte Carlo analysis, a sensitivity analysis was performed to assess the uncertainty in total volume and PRP volume inputs on the overall cost allocation. When the 25th percentile and 75th percentile disposal volumes were used instead of the median disposal volume, each PRP's overall remedial cost allocation changed by no more than 2.5% (data not shown). The sensitivity analysis demonstrated that while the overall method had some uncertainty, the final cost allocation is relatively robust and does not change widely when lower-end and higher-end volume assumptions are used uniformly for all PRPs. An advantage of the Monte Carlo approach is that alternative sensitivity analyses (e.g., varying each PRP's contribution within a range, where some PRPs may contribute at the low end of the range and others contribute at the high end of the range) could be performed if warranted based on site-specific conditions or directives.

Application of the method by the courts

U.S. district court decision

In its *Findings of Fact and Conclusions of Law* (Lyondell v. Albemarle, 2007), the U.S. district court largely adopted the allocation methodology described herein, electing only to alter some inputs to the calculations as further described below. The concept of risk magnitude and its relationship to the site's remediation cost was fully adopted by the court. The court also found that the RDCs identified through the risk magnitude and detection frequency analysis provided a "practicable, scientifically sound means to allocate responsibility based on relative toxicity," while recognizing that it may not capture 100% of the risk. However, the court elected not to attempt to quantify volumes of waste streams specific to each RDC, and instead opted to tie RDCs originating from a particular PRP to the PRP's aggregate volume disposed at the site.

For the waste volume estimates, the court found the Monte Carlo methodology described herein was superior to the approaches offered by the PRPs due to its ability to better account for the inherent uncertainty in the available historical evidence. The court opted, however, to utilize different volume inputs for some PRPs within its own Monte Carlo analysis, stemming from the court's "greater willingness to consider the testimony and trip tickets rather than only the more technical documentary evidence that Dr. Newell, an engineer, favored." These divergences in how the court elected to apply the overall methodology resulted in a different final allocation than summarized in Table 3. The court also elected to adjust final allocation share to account for cooperation and care factors (Lyondell v. Albemarle, 2007).

U.S. court of appeals decision

Following the decision by the U.S. district court, several PRPs appealed to the U.S. court of appeals contesting the district court's equitable allocation of the site's costs. The appeal, as it related to the subject of this paper, was based on several arguments regarding the use of Monte Carlo methods to estimate disposal volumes. Specifically, the appealing PRPs claimed:

1. the Monte Carlo method had not been peer reviewed as applied to CERCLA allocations;
2. it was not generally accepted for use in CERCLA allocation;
3. it was developed specifically for use in this litigation;
4. it has not been tested as applied to CERCLA allocation and has a rate of error that cannot be evaluated; and
5. it is not relevant because it is "equivocal."

The appellate court found no error with respect to the use of Monte Carlo analysis in the allocation methodology, recognizing the utility of the method when an exact solution is infeasible, but the data provide a known range (Lyondell v. Occidental, 2010). The court pointed to the wide use of Monte Carlo analysis in the physical sciences and finance, as well as its endorsement by USEPA (1997) for very similar applications, as a basis for rejecting the claims above. The court further concluded that, "just because a Monte Carlo simulation produces a range of outcomes, rather than one single numerical value, does not mean it is speculative. If anything, Monte Carlo analysis provides greater certainty than the basic alternatives: using one of the three data points or using the arithmetic average of all three." The precedent established by this case, particularly the use of Monte Carlo analyses for CERCLA allocation, was recognized as one of the "Top 10 Expert Rulings of 2010" (IMS 2010).

Application of the approach at other sites

The allocation approach described herein utilized tools and data analysis approaches that are widely accepted within the scientific community and combined them in a new way to derive a fair and equitable allocation for the site-specific conditions. The method can be adapted to include more or less rigor for individual components within the analysis as warranted based on project-specific conditions, objectives, and constraints. The methodology is not intended to be a "one-size-fits-all" or a prescriptive approach to the complex task of cost allocation, as site-specific conditions and directives will nearly always necessitate adjustments to any allocation framework. However, the authors believe the fundamental aspects of Risk Magnitude, Monte Carlo analysis, and use of GIS, as generally described herein, either on their own or in combination, can be applied to nearly any site where allocating costs of remediating complex mixtures of chemicals in environmental media (i.e., soil, groundwater, sediments) is the objective. The application of the methodology by a federal court provides a sound basis for its consideration as part of future allocation projects.

Declaration of interest statement

No potential conflict of interest was reported by the authors.

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