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Review of Ground-Water Quality Monitoring Network Design

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Abstract: Ground-water quality monitoring network design is defined as the selection of sampling sites and (temporal) sampling frequency to determine physical, chemical, and biological properties of ground water. The main approaches to ground-water quality monitoring network design were identified as hydrogeologic and statistical. The various methods for network design available in the hydrologic literature have been evaluated by considering the spatial scale of the monitoring program, the objective of sampling, data requirements, temporal effects, and range of applicability. Considerable advance has been made over the last two decades that now permit the application of methodical and testable approaches to ground-water quality monitoring network design, although they mostly serve for preliminary analysis and design. The opinion of the Task Committee on Ground-Water Quality Monitoring Network Design is that as there continues to be advances in hydrogeochemistry, ground-water hydrology, and risk and geostatistical analysis, methods for ground-water quality monitoring network design will be improved and refined, and they will become ever more useful in the important mission of environmental protection.

Introduction

Ground-water quality monitoring network design is defined as the selection of sampling points and (temporal) sampling frequency to determine physical, chemical, and biological characteristics of ground water. The modern concept of ground-water quality monitoring network design requires that consideration be given to: (1) The spatial and temporal coverage of the sampling sites; (2) the competing objectives of a monitoring program; (3) the complex nature of geologic, hydrologic, and other environmental factors; (4) the uncertainty about parameters (geologic, hydrologic, and environmental) needed in the design process; and (5) the range of applicability of the various methods for network design.

The interest in ground-water quality monitoring has increased significantly in the United States since the mid-1970s, fueled primarily by federal leg-

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islation that includes the Clean Water Act, the Safe Drinking Water Act, the Toxic Substances Control Act, the Resource Conservation Recovery Act (RCRA), and the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA or "Superfund"). Legislative enactment of environmental regulatory policy is a reflection of increased public interest in environmental quality issues, and in ground-water quality preservation in particular. Longstanding interest in ground-water quality monitoring has existed also on the part of federal agencies, most prominently the United States Geological Survey, as well as state and local agencies of which statutory mandates require the surveillance of the quality of ground water. Against this background of heightened interest on the part of public agencies, private organizations, and at last certain sectors of the general public, there has been a steady increase in the number, and quality, of technical articles related to ground-water quality network design in the hydrological literature. Today, there exist methodologies for ground-water quality network design, something that has been made possible to a certain extent by the advances in analytical, numerical and statistical modeling of ground-water processes achieved over the last two decades.

Under the auspices of the Committee on Probabilistic Approaches to Hydraulics of the Hydraulics Division of the American Society of Civil Engineers, the Task Committee on Ground-Water Quality Monitoring Network Design has prepared this review paper. The task committee's objective was to survey the technical literature in the field of ground-water quality monitoring network design and to identify and evaluate the most promising methods for network design. This effort begins with a discussion of multiple objectives, spatial scales, and hydrogeologic considerations, in ground-water quality monitoring network design. The other section of this paper contains a survey of the main methodologies for ground-water quality monitoring network design, followed by a summary and conclusions.

Multiple Objectives, Spatial Scales and Hydrogeologic Considerations

Multiple Objectives

The objective of a ground-water quality monitoring activity is the main factor determining the cost, the level of detail, and the appropriate method for the planning of a monitoring network. A ground-water quality monitoring program typically has one, or more, of the following objectives: ambient monitoring, detection monitoring, compliance monitoring, or research monitoring (Todd et al., 1976). Ambient monitoring establishes an understanding of characteristic regional ground-water quality variations over time. This type of monitoring is accomplished through routine sampling of wells on a regional basis. The wells sampled are often public water supply, industrial, or domestic wells, rather than special monitoring wells. Ambient ground-water quality monitoring is exemplified by the national water collection program supported by the United States Geological Survey [see "Groundwater" (1980)] or the well-head protection programs of the U.S. Environmental Protection Agency (EPA). Detection monitoring has the primary function of identifying the presence of targeted contaminants as soon as their concentrations exceed background or established levels. This type of monitoring is required at and around point and nonpoint sources of ground-water contaminants. A typical example is the placement of downgradient and upgradient monitoring wells in the vicinity of a toxic-waste landfill. Compliance monitoring denotes a stringent set of ground-water quality monitoring requirements for chemical compounds at a disposal facility after detecting their presence in monitoring wells. Compliance monitoring is enforced to verify the progress and success of ground-water cleanup and remediation works. Research monitoring consists of the detailed spatial and temporal ground-water quality sampling tailored to meet specific research goals [see Mackay et al. (1986)].

Given a monitoring objective, one must select a methodology to design the sampling network. Some methods for ground-water quality network design result in mathematical models that, upon solution, yield sampling locations (i.e., the network) and frequency of sampling times if there is no fixed sampling interval. Such mathematical models typically seek to optimize a specific criterion of performance, or objective function. One of the key difficulties in the design of ground-water monitoring networks via mathematical models is to choose objective functions that faithfully represent a monitoring objective. There are two main reasons for this: (1) The existence of competing criteria that could express, at least partly, the monitoring objective; and (2) the dynamic nature of ground-water quality monitoring, with possible changes of the monitoring objective over time, e.g., from detection to compliance and back to detection (Sanders 1987; Hirsch et al. 1988; Reichard and Evans 1989). The analytical complexities introduced by competing criteria are well known in the hydrologic and water resources literature (Cohon 1978). As an example, in an ambient monitoring program one could consider several plausible objective functions, including the cost of monitoring (Raucher 1986; Massmann and Freeze 1987a; Reichard and Evans 1989; Schwarz and Byer 1984), the accuracy of contaminant concentration predictions (Rouhani 1986; Loaiciga 1989), severity of exposure (Hsueh and Rajagopal 1988), economic impacts associated with ground-water monitoring (Sharefkin et al. 1984), risk quantification associated with ground-water pollution [*Benefits* (1983); Hobbs et al. (1988); and see also *Risk* (1983) for an exposition of risk assessment and management), or a combination of different functions (Andricevic 1990).

Regarding the choice of suitable objective functions, they can be classified as either ultimate objectives or surrogate objectives. Ultimate objectives explicitly consider the value of ground-water quality information in achieving monitoring goals such as environmental protection (Hsueh and Rajagopal 1988), the reduction of remediation costs (Reichard and Evans 1989), and minimizing exposure risks (Massmann and Freeze 1987a) or health hazards. Surrogate objectives, on the other hand, are typically substitutes for ultimate objectives. They are used in many cases to bypass the difficulties posed by the formulation of network design problems in terms of ultimate objectives. Examples of surrogate objectives are the minimization of statistical parameters such as the variance of contaminant concentrations (Rouhani 1985) or the minimization of the maximum absolute deviation between actual and predicted variables (Loaiciga and Church 1990). The use of surrogate objectives often leads to more tractable monitoring-network design problems than those based on ultimate objectives. They are, however, only approximations, and the results derived from surrogate objectives must be interpreted accordingly. With the continuing technological and theoretical advances in hydrogeology and decision-making modeling, the state of the art in ground-water quality monitoring network design will most likely rely more heavily on methods based on ultimate objectives.

Another important consideration in network design is the dynamic nature of ground-water quality programs. In many instances, the network design

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depends upon the spatial (and also temporal) distribution of chemical concentrations in the subsurface, which are often unknown before sampling is undertaken. Therefore, network design is typically an iterative process, whereby the sampling program (in space and time) must be revised and updated in response to changes in information needs and in the gathered data. Examples of such iterative schemes for ground-water quality monitoring network design are given in the sequential variance-reduction method by Rouhani (1985) and Graham and McLaughlin (1989a,b), and in the probability-based approaches of Rouhani and Hall (1988) and Scheibe and Lettenmaier (1989). The selection of objective function in ground-water quality monitoring network design is, therefore, linked to the dynamic nature of the data collection effort and to the ultimate use to which information is to be devoted. Within this evolving decision-making environment, one must weigh, in addition, the trade-offs that arise when choosing among surrogate objectives and ultimate objectives [see Schwarz and Byer (1984) and *Benefits* (1983) for the case of ultimate objectives in which future remediation, water supply, or health costs are balanced against monitoring costs].

In summary, the choice of a criterion of performance, or objective function, is perhaps the most important step in the design of ground-water quality monitoring networks. (In fact, as will be seen later in this paper, some of the leading methods in network design are classified according to the type of objective function.) Although there is not a single best criterion, the selected criterion must be intimately linked to the overall monitoring objective (ambient, compliance, detection, or research), and it must reflect the trade-offs between competing functions (e.g., exposure assessment or prediction of concentration) and account for the uncertainties commonly plaguing the data used in network design.

Spatial Scales

The objective of a ground-water quality monitoring program determines the spatial scale of the sampling network. The spatial scale defines the areal coverage of the monitoring network. For example, ambient monitoring is often synonymous with regional ground-water quality monitoring. At the other extreme, compliance monitoring implies site-specific coverage. The spatial scale has profound effects on the choice of objective functions (or performance criterion), on the types of hydrogeologic data and models needed in the process of network design, and on the (temporal) sampling frequency. To illustrate, a regional-scale monitoring program typically requires annual or semi-annual sampling frequency and emphasizes the geographical distribution of sampling wells relative to population or economic centers (Hsueh and Rajagopal 1988). In contrast, a compliance monitoring program most likely will dictate monthly, or at most quarterly, sampling frequency, rely heavily on local-scale hydrogeologic features, and require detailed information on aquifer properties (e.g, hydraulic conductivity) to best characterize the migration paths of the contaminant plumes (*RCRA* 1986; Meyer and Brill 1988). It will become apparent later in this review that the choice of objective function, the spatial scale of the monitoring network, and hydrogeologic considerations are inseparable, and, together they determine the best method for each particular application to network design.

Hydrogeologic Considerations

Hydrogeologic considerations provide the fundamental data and rationale needed to design any ground-water monitoring network. Without adequate hydrologic characterization, one stands little chance of implementing a successful, cost-efficient monitoring plan. This is due primarily to the complexity inherent to the subsurface. Complicated spatial patterns in aquifer properties often cause contaminants to migrate in unexpected directions along preferential flow paths such as buried fluvial channels or fracture zones. Preferential flow paths can also exist along sewer lines or well casings.

Details of heterogeneous aquifers can be mapped directly only if very closely spaced well data are available. Since this is rarely the case, we are typically faced with a problem of using sparse data to estimate the unknown spatial pattern in aquifer properties. Intuitively, estimating an unknown spatial pattern is impossible unless we know something about the processes that created the pattern. In the subsurface, these processes are geologic, and thus geologic characterization can provide invaluable information on the spatial distribution of aquifer properties. The geologic systems responsible for creating the aquifers are often regional in scale (Kaiser et al. 1978; Fogg et al. 1983), and therefore one must examine the regional geology to determine the geologic processes that deposited the system. Furthermore, ground-water circulation is often controlled by regional and topographic features, including major rivers, outcroppings of the aquifer, and vertical leakage across confining units. Site-specific analysis alone tends to miss such features, leading to a less than comprehensive understanding of the local system.

Regional Characterization

Geology and Heterogeneity. Hydrogeologic investigations have traditionally made use of available information on the regional stratigraphy, structural setting (e.g., locations of major faults), and general characteristics of the rocks or sediments. Such information is essential, but it tends to paint an oversimplified picture of the regional heterogeneity and often leads to the assumption of a layered-cake arrangement of laterally continuous aquifers and aquitards. In reality, most aquifer systems are laterally discontinuous—even on the regional scale. For example, Fig. 1 shows a sand-percent map for the Wilcox Group in the East Texas Basin (the "sands" consist of elongated bodies approximately 1–30 m thick, with a total formation thickness ranging from 300 m to 1,000 m; the map was constructed by measuring cumulative sand and formation thicknesses on borehole resistivity logs). The most transmissive parts of the aquifer system are delineated by the high sand-percent zones, which form a dendritic pattern that partly reflects the fluvial mode of deposition. Many of the low sand-percent zones behave as aquitards because these zones lack high-permeability, laterally continuous sands. Thus, average ground-water velocities, as well as appropriate monitoring strategies, will differ substantially across this system.

Though the sand-percent map in Fig. 1 spans an entire basin, the same kinds of maps can be constructed on subregional scales that would more readily tie into local-scale studies. Consultants seldom have the time and resources to perform regional studies, but regional geologic and hydrogeologic characterizations are frequently available in the literature of state geological surveys and the U.S. Geological Survey. For more information on characterization of regional geologic heterogeneity, the reader can refer

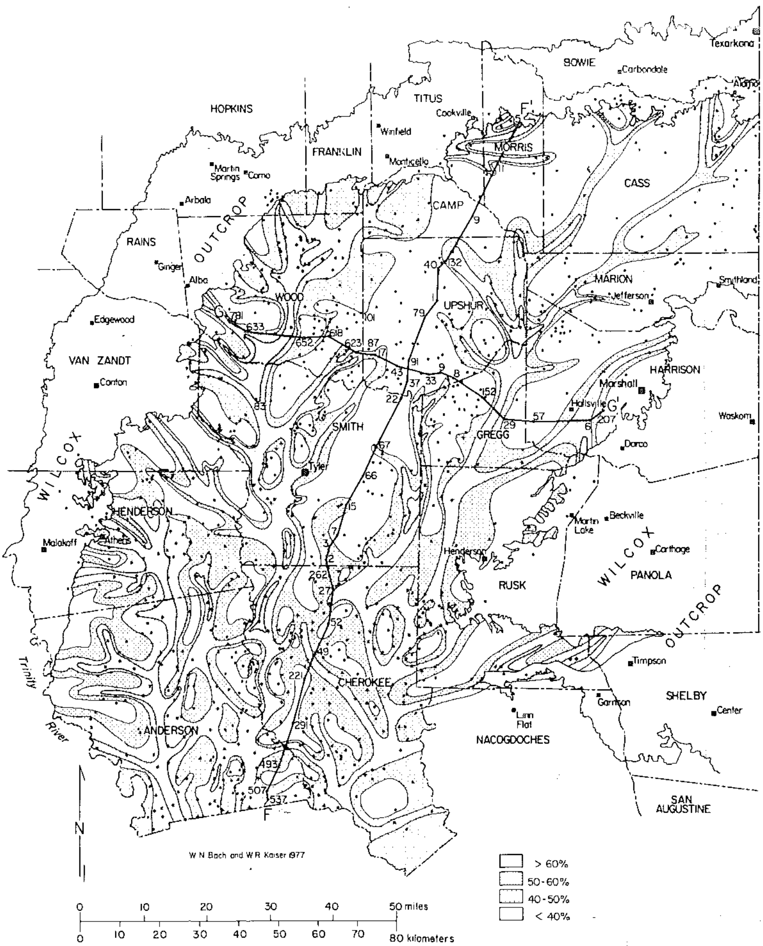

FIG. 1. Sand-Percent Map, Wilcox Group, East Texas Basin [from Kaiser et al. (1978)]

to Galloway and Hobday (1983), Reading (1978), and Selley (1978), among others.

Ground-Water Circulation. Mapping of hydraulic gradients requires data on water levels (hydraulic head) in wells collected at different times and depths. The quantity of these data is typically inadequate on a local scale to construct reliable hydraulic gradient maps. In that case, the best recourse might be to construct regional head maps, which generally gives one the opportunity to utilize a much more complete data set. Furthermore, in strongly anisotropic aquifer systems, the hydraulic head or potentiometric head map will not indicate true ground-water flow directions except in areas where the hydraulic gradient happens to be colinear with one of the principal axes of hydraulic conductivity.

Vertical ground-water flow, which can be critical to issues of contaminant transport and ground-water monitoring, is often neglected in the analysis of ground-water circulation. Vertical hydraulic gradients are important because: (1) They influence the direction of contaminant movement; and (2) they may be larger than horizontal gradients by a factor of 100 or more in horizontally stratified media. The vertical hydraulic gradients tend to be high because horizontal stratification creates low effective vertical hydraulic conductivities K_v compared to the effective horizontal hydraulic conductivity K_h. (Interbedded sands, silts, and clays typically yield values of the ratio $K_h/K_v > 10^3$.) However, due to the low K_v, the large vertical hydraulic gradients do not necessarily indicate high rates of vertical flow. Only when local vertical pathways of higher K_v (e.g., faults or fractures, zones of good vertical interconnection of sands, or improperly cased boreholes) exist in the presence of strong regional vertical gradients can substantial vertical flow occur. This phenomenon can cause extreme differences between predicted and actual contaminant migration pathways (Huyakorn et al. 1983).

The regional investigation generally cannot define local zones of intense vertical flow, but it can often estimate or map the prevailing vertical hydraulic gradients. Then, if in a site investigation vertical gradients are found to be much lower than in the surrounding region, one can conclude that aquifers at the site are well connected vertically and that relatively rapid vertical flow may be occurring.

Water Chemistry. Characterization of the regional ground-water chemistry serves three purposes. First, determination of the natural evolution of hydrochemistry from recharge area to discharge area provides information on the rock-water reactions occurring in the system. Such information can in turn provide important clues to geochemical and biological attenuation of contaminants. Moreover, in certain fractured rock systems, the geochemical data may more readily lead to hydrologic interpretation of subsurface flow systems than piezometric data. Second, changes in the hydrochemistry can be used as an environmental tracer that reflects the gross flow paths in the system. For example, ground-water chemistry in many sandstone aquifers evolves from a low pH, calcium-bicarbonate water type in recharge areas to a high pH, sodium-bicarbonate water at some distance downgradient. Knowledge of systematic evolution in ground-water chemistry can be used as a basis for detection or mapping recharge and discharge areas. Third, to assess the seriousness of inorganic (and, at time, organic) chemical contamination at a site, one must have data on the background or regional water chemistry (Loaiciga 1990).

Developing a Conceptual Model. A conceptual model of the groundwater system hydrology should be the final product of a regional investithe geology, ground-water hydraulics, and water chemistry. Whether formulated in strictly conceptual terms or translated into a numerical groundwater model, the ground-water model is indispensable in implementing several of the methods for ground-water quality monitoring network design to be discussed in the following. This model should elucidate the principal mechanisms of recharge, discharge, migration, and rock-water interactions that are so vital in designing a ground-water quality monitoring network.

Site-Specific Characterization (Local Scale)

Geology and Heterogeneity. Potential contaminant migration pathways are controlled primarily by site geology and its influence on aquifer parameters, e.g., hydraulic conductivity, and by the setting of the site within the regional flow system. Thus, an adequate site-specific characterization of

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geology and heterogeneity is indispensable in developing a successful monitoring network plan.

In the past, hydrogeologists generally considered heterogeneity too complex to characterize at a site. This is still true if one attempts to use only the available measurements of hydraulic conductivity, retardation coefficients, etc., but in the case of sedimentary clastic aquifers, which comprise the bulk of the world's ground-water resources, advancements in geologic depositional models and in stochastic/geostatistical methods makes characterization of site heterogeneity more feasible than ever before. The latest depositional models provide the basis for estimating the type of interwell heterogeneity, and stochastic/geostatistical methods such as conditional simulation (Journel and Huijbregts 1978) provide a means for dealing with the uncertainty inherent to estimation of interwell heterogeneity (Fogg 1989).

Successful aquifer description requires an integrated approach comprised of at least five tasks as shown in Fig. 2: (1) Geologic characterization; (2) borehole and surface geophysics; (3) measurement of hydraulic properties; (4) stochastic or deterministic estimation of interwell patterns of heterogeneity; and (5) modeling of flow and transport. State-of-the-art execution of each task may require several specialists. Well-trained hydrogeologists are capable of ascertaining the general geology, measuring hydraulic properties, and modeling the system. However, a complete understanding of hydrologic and geologic processes and of their spatial variability and interaction can be facilitated by having the expert opinion of geostatisticians (to help estimate the interwell geologic patterns and quantify spatial aquifer variability), geophysicists (to interpret geophysical data), and, possibly, sedimentologists (to interpret the depositional environments and to form predictive geologic models). This type of interdisciplinary approach is essential in developing a sound data base to initiate the implementation of methods for ground-water quality monitoring network design.

Vadose Zone Characterization

The vadose zone is that region of the geologic profile that extends from the land surface to the water table. The vadose zone is important to ground-

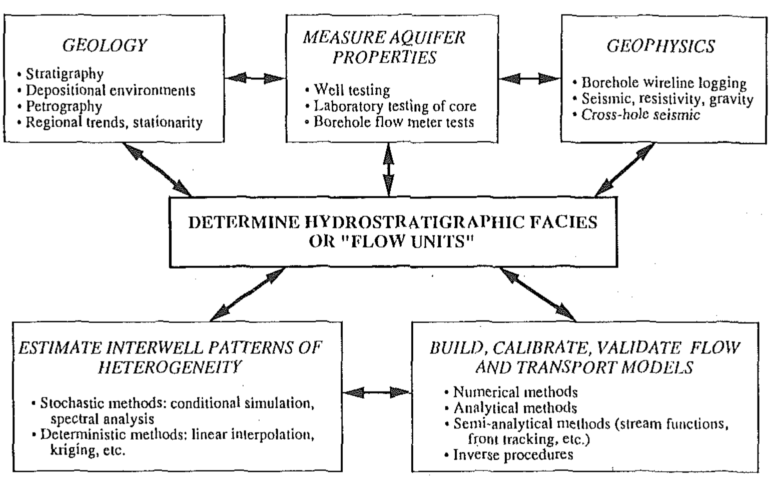

FIG. 2. Chart Showing Components of Integrated Aquifer Description

water quality monitoring, since, typically, contaminants applied at the surface migrate through the vadose zone before being advected and dispersed by ground-water flow. The behavior and fate of chemicals in the vadose zone is rather complex (Everett et al. 1982), and it influences the subsequent spatial and temporal evolution of contaminants in ground water. The influence of soil-water content, saturated and unsaturated hydraulic conductivity, matric potential (or soil tension), and soil characteristics in downward contaminant migration is incorporated in some network design methods (Everett 1984), although most of them either neglect or consider the vadose zone only indirectly (e.g., as a delay in contaminant transport). These methods for ground-water quality monitoring network design will be discussed in the next section.

This discussion has laid the groundwork for a systematic classification of approaches to ground-water quality monitoring network design. The reader is reminded that our classification of network design methods intends to elucidate similarities, as well as expose contrasts, among various alternative methods. However, there is always an element of arbitrariness in any classifying scheme, and network design can be the product of a combination of methods.

Methodologies for Ground-Water Quality Monitoring Network Design

This section reviews alternative approaches to ground-water quality monitoring network design, in which the principles underpinning each technique are evaluated. There are two general types of approaches to network design, namely the hydrogeologic and the statistical approaches, where the latter type can be further divided into simulation, variance-based, and probability-based techniques. To compare relative advantages and disadvantages of each of these approaches, one should consider many factors, including: (1) The scale of the monitoring program (i.e., field-scale or regional-scale); (2) the objective of the monitoring program (ambient monitoring, detection monitoring, compliance monitoring, or research monitoring); (3) the type of available data (hydrogeologic, geologic, ground water quality, etc.); (4) the nature of the investigated subsurface process (vadose zone or saturated-zone contamination); (5) the steady-state versus transient nature of ground-water quality properties; (6) the resources available to accomplish the monitoring program; and finally (7) the fact that ground-water quality monitoring network design is an iterative process. An initial design might be upgraded as more data and resources become available. The approaches discussed in this paper are intended to provide a preliminary design for the layout of sampling sites and selection of sampling frequency. The ultimate selection of a sampling plan is often influenced, in many cases, by institutional, legal, and site-specific considerations.

Environmental monitoring programs are dynamic in nature and phased in their implementation. For example, the ground-water monitoring document prepared by the U.S. EPA (*RCRA* 1986), indicates that

> . . . surface geophysical techniques can be effectively used in tandem with the installation of monitoring wells as a first phase in the assessment program to obtain a rough outline of the contaminant plume. Based on these findings, a sampling program may subsequently be undertaken to more clearly define the three-dimensional limits of the

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contaminant plume. In the third phase, a sampling program to determine the concentration of hazardous waste constituents in the interior of the plume may be undertaken. . . .

The reader should keep this step-by-step approach in mind in order to appreciate the utility and limitations of the approaches and methodologies reviewed in this work.

Hydrogeologic Approach

The hydrogeologic approach (Everett 1980; *RCRA* 1986) is the basis of the procedures most commonly used in practice. As the title of this approach indicates, it is based on qualitative and quantitative hydrogeologic information. Admittedly, the science of hydrogeology encompasses all approaches that deal with ground water, including various statistical approaches. All the methods discussed in this paper could therefore be categorized as hydrogeological. Nevertheless, to be consistent with previous work on subsurface monitoring, the term "hydrogeological approach" will be used to describe the case where the network is designed based on the calculations and judgment of the hydrogeologist without the use of advanced geostatistical methods. More specifically, the number and locations of sampling sites (i.e., wells) are strictly determined by the hydrogeologic conditions near the source of contamination such as a waste impoundment. As an example, the RCRA guidelines for ground-water monitoring (*RCRA* 1968) require, at a minimum, four ground-water monitoring wells: one well upgradient and three wells downgradient from the source of contaminants. These guidelines establish criteria for well siting and specify that upgradient wells must be: (1) Located beyond the upgradient extent of potential contamination from the hazardous waste management unit to provide samples representative of background water quality; (2) screened at the same stratigraphic horizon(s) as the downgradient wells to ensure comparability of data; and (3) of sufficient number to account for heterogeneity in background ground-water quality. The placement of downgradient wells must consider: (1) The distance to the contaminant source and the direction for ground-water flow; (2) the likelihood of intercepting potential pathways of contaminant migration; and (3) the physical and chemical characteristics of the contaminant source controlling the movement and distribution of contamination in the aquifer.

The hydrogeologic approach is better suited for site-specific studies where there is a well-delimited source of contamination. The geological facies, structural geologic features, and the local and regional ground-water flow patterns at the site determine the spatial (horizontal and vertical) distribution of sampling points. The main objective of the hydrogeologic approach is to detect pollution as soon as the contaminant plume leaves the confinements of a waste-impoundment site. Geologic features such as aquifer layering and the presence of fractures determine the need for the vertical placement of sampling points. These points are in the form of either multiscreened wells or well clusters.

Fig. 3 shows an example of the hydrogeologic approach to ground-water quality monitoring network design. The aquifer is a glacial outwash with two sand layers in hydraulic connection through a low-conductivity, glacial till aquitard. Bedrock is highly impermeable granite. The upper sand layer has a relatively high hydraulic conductivity and greater Darcian velocity with a southerly direction. The lower, more compacted, sand layer has a

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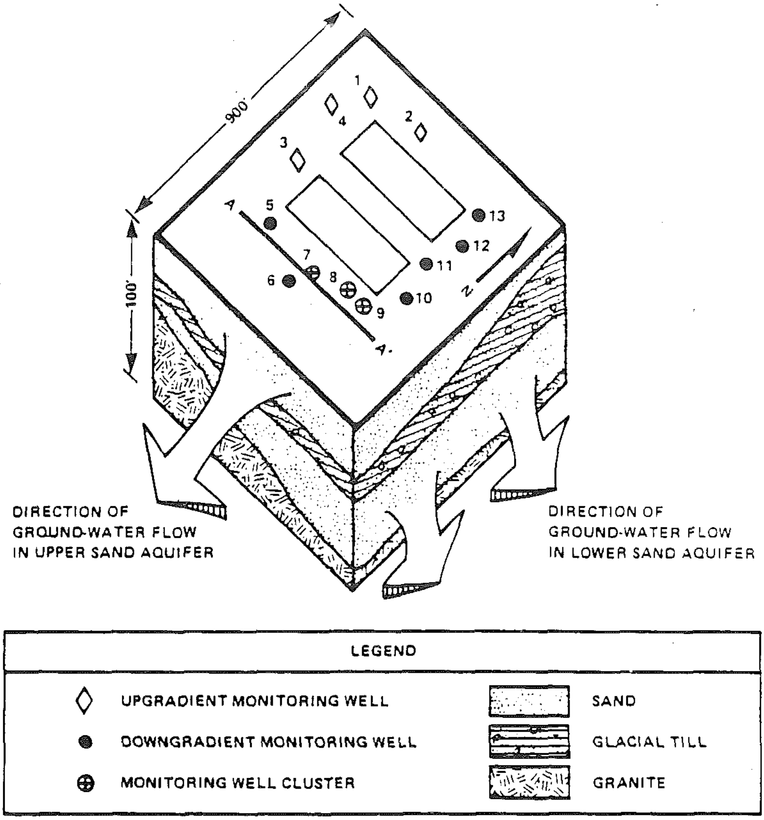

FIG. 3. Placement of Monitoring Wells in Hydrogeologic Approach [from *RCRA* (1986)]

lower hydraulic conductivity, and its ground water is part of a deeper regional flow regime with ground-water velocity having a southeasterly direction as indicated by the arrows. Flow through the glacial till aquitard is downward leakage. There are two lined waste impoundments to be located in the upper sand layer with a depth not exceeding the water table elevation in the upper unconfined aquifer. Further information includes the stratigraphic features of the various formations as well as piezometric level contours for the upper (unconfined) and lower (confined) aquifers. Based on this information, it is proposed to install both down- and upgradient wells. The downgradient wells are located in the southern and the eastern perimeters of the impoundments in order to determine the ground-water quality downgradient of the contaminant source. The upper, more permeable sand layer is tapped by five monitoring wells, three of which are well clusters that penetrate the various formations (upper sand, till, and lower sand). Any leachate must pass the upper sand and till layers before reaching the lower sand layer. Therefore, fewer single wells are required along the eastern perimeter, as shown in Fig. 3. The upgradient monitoring wells are located in the northern and western perimeters, and are screened in the two sand layers to provide background water quality information to be contrasted with the downgradient wells.

This example illustrates that the hydrogeologic approach is data-intensive, tailored for site-specific applications such as at RCRA sites, where the primary objective is the early detection of contaminant migration beyond the site boundaries. The hydrogeologic approach can also yield information on sampling frequency, and this is typically done by an elementary application of Darcy's law to describe ground-water velocity near a well and considering the well diameter [see *Statistical* (1989)]. Table 1 lists sampling intervals found in common situations [adapted from *Statistical* (1989)].

A shortcoming of the hydrogeologic method recommended by the United States Environmental Protection Agency (*RCRA* 1986) is its lack of attention to vadose transport. Everett et al. (1984) recommended early-alert monitoring strategies close to the contaminant sources in the vadose zone as part of any ground-water monitoring program.

Statistical Approach

The presented hydrogeologic approach relies heavily on descriptive information about the aquifers of interest, and often does not fully utilize the available quantitative hydrogeologic information. This is partially due to the inherent uncertainties in the results of quantitative models of ground-water flow and mass transport. These uncertainties stem from many sources, such as: (1) The ever-present simplifications and errors in both the modeling stage and the numerical/analytical solution phase; and (2) inherent heterogeneities of the involved variables and parameters. Dettinger and Wilson (1981) denote the first source as the information uncertainty, which represents the lack, in quantity or quality, of information concerning the aquifer system. This uncertainty may be reduced by various strategies, notably further measurements and analysis. The second source of uncertainties arises from the spatial and temporal variations of parameters such as recharge rate, hydraulic conductivity, and dispersion coefficients. These variations are extremely complicated and in general defy exact description. Some researchers proposed to model such phenomena as classical random variables with no spatial structures, which yields monitoring plans based on random sampling and related procedures ("Test" 1986). The assumption of

TABLE 1. Typical Ground-Water Quality Sampling Intervals [from *Statistical Analysis* (1989)]

Unit (1)	K_h[a] (ft/day) (2)	Ne[b] (%) (3)	V_h[c] (in./mo) (4)	Sampling interval (5)
Gravel	10^4	19	9.6×10^4	Daily
Sand	10^2	22	8.3×10^2	Daily
Silty sand	10	14	1.3×10^2	Weekly
Till	10^{-3}	2	9.1×10^{-2}	Monthly[d]
Sandstone	1	6	30	Weekly
Basalt	10^{-1}	8	2.28	Monthly[d]

[a]Horizontal hydraulic conductivity, in feet per day (ft/day), 1 ft/day = 3.53×10^{-4} cm/s.
[b]Effective porosity.
[c]Average horizontal velocity in inches per month (in./mo), 1 in./mo = 9.80×10^{-7} cm/s. Based on a hydraulic gradient of 0.005.
[d]Use a monthly sampling interval or an alternate sampling procedure.

no spatial structure, however, turns out to be inappropriate for a large number of involved ground-water variables, which clearly exhibit spatial tendencies. As a result, a new class of stochastic techniques have been developed to analyze ground-water flow and transport processes, based on spatially distributed random variables. The pioneering work in stochastic ground-water hydrology was conducted by Freeze (1975), who viewed log-transmissivity as a spatially distributed variable that yielded piezometric head estimates with given accuracies. This led to an extensive collection of papers on statistical modeling of ground-water flow, e.g., Smith and Freeze (1979a, 1979b), Dagan (1982a, 1982b, 1982c; 1985a, 1985b), Hoeksema and Kitanidis (1984, 1985), Kitanidis and Vomvoris (1983); Rubin and Dagan (1987, 1988), and Dagan and Rubin (1988). The statistical approach was later applied to the transport equation by such authors as Dagan (1982, 1984, 1988), Gelhar and Axness (1983), Neuman et al. (1987), and Rubin (1990). These authors, among others, recognized the fact that the spatial and the temporal variability of ground-water properties are highly complex, and, therefore, the statistical characterization of their variations has, by necessity, been the main tool in their analyses of transport phenomena.

One of the most common tools used in stochastic ground-water analyses is geostatistics, which can be described as a systematic approach to making inferences about quantities that vary in space. Such a quantity, which is a function of spatial coordinates, may be represented as $z(x_1)$ $z(x_1,x_2)$, or $z(x_1,x_2,x_3)$, depending on whether it varies in one, two, or three dimensions, respectively. As noted by the ASCE ("Review" 1990a), ground-water hydrologists rely heavily on site-specific data to describe the geometry and properties of formations of interest. However, in the majority of cases, the spatial variability of such properties precludes error-free (i.e., "deterministic") predictions, even when extensive measurements are available. Samples or measurements taken at different locations within the same aquifer display a wide range of variability. As a consequence, one cannot predict with certainty the value of the head or of the concentration at a well, even if these quantities have been measured at other nearby wells.

In geostatistics, the variable property is viewed as a spatial function presenting complex variations that cannot be described effectively either by simple deterministic functions, such as polynomials, or by borrowing directly from classical statistics. Such a function is the subject of geostatistics, and has been called a regionalized variable by Matheron (1971) and Journel and Huijbregts (1978). Examples of regionalized variables are hydraulic conductivity, which usually displays spatial variability due to the geology of the aquifer, or the concentration of a natural chemical compound in ground water, which varies in both space and time.

Geostatistics proposes to view a regionalized variable as one of many possible realizations of a random field (a spatial random function). It should be noted here that the decision to select random functions to model a regionalized variable is only a matter of analytical convenience. This does not indicate that the phenomenon under study is indeed random; instead it simply implies that when one lacks sufficient data to make deterministic inferences, it is reasonable to resort to probabilistic tools.

In many instances, it is useful to write

$$Z(x) = m(x) + \varepsilon(x) \quad \ldots\ldots (1)$$

where $Z(x)$ = the random variable representing $z(x)$; $z(x)$ = the aquifer

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property of interest at x; $m(x)$ = mean value of $Z(x)$, also known as drift or trend; and $\varepsilon(x)$ = fluctuation value at x.

The mean represents a trend that is a deterministic and usually smooth function of space which accounts for the part of $Z(x)$ that represents its large-scale variations. The fluctuation is superimposed on the trend to sum up to the point value of $Z(x)$, characterizing the usually small-scale erratic variations of the regionalized variable. This modeling approach enables us to describe unexpected variations of ground-water properties when one moves from one measurement point to another.

The fluctuation term $\varepsilon(x)$ is assumed to oscillate about a zero value and exhibits a spatial statistical dependence. The existence of this dependence is intuitively justified by the fact that values sampled at neighboring locations tend to be more alike (numerically) than those values collected at distant locations. The incorporation of this spatial correlation is an important characteristic of geostatistical analyses. The correlation structures are usually represented by covariance and/or semivariogram functions.

The geostatistical framework permits us to perform many tasks, including: (1) Calculating the most accurate (according to well-defined criteria) predictions, based on measurements and other relevant information; (2) quantifying the accuracy of these predictions; and (3) performing simulation by generating spatial functions or sets of values of a given property that are statistically consistent with available information. The noted measure of accuracy and the capability of generating an unlimited number of simulations have led to the development of various geostatistical techniques for ground-water quality monitoring network design that may be grouped into three classes: simulation, variance-based, and risk-based approaches. These techniques are discussed in more detail in the following sections. For a more in-depth review of geostatistics in hydrogeology, readers are referred to the ASCE papers ("Review" 1990a, 1990b).

Simulation Approach

This approach primarily uses the simulation capabilities of the geostatistical models, and is a versatile technique that is gaining popularity in the broad field of ground-water quality management [e.g., see Wagner and Gorelick (1987, 1989)]. In these works, the hydraulic conductivity of the porous medium is viewed as a regionalized variable. Due to the significant influence of hydraulic conductivity on the ground-water velocity distribution and, thus, on the advection and dispersion of chemical compounds in ground water, both velocity and pollutant distributions may be also viewed as regionalized variables. In simulation works for fractured formations, the fracture density, the aperture size, and the length, location, and orientation of the fracture are generated as regionalized variables (Long and Billaux 1987; Moreno et al. 1988; Tsang et al. 1988).

The conceptual backbone of the simulation approach is that by generating multiple synthetic fields of hydraulic conductivity, for each of which there will be a corresponding contaminant field, it is possible to determine the statistical properties of mass transport in an aquifer, and thus the reliability of a monitoring network. For any given arrangement of monitoring wells and sampling frequency, the simulation approach yields important, quantities such as the probability that a contaminant plume might miss all of the sampling points an go undetected, i.e., the probability of a false negative. By careful experimental design of the computer simulation, various network configurations can be entertained and examined for their adequacy in contaminant detection. Therefore, there is some leverage for optimization in the simulation approach. Examples of simulation applications in ground-water quality monitoring network design can be found in such works as Massmann and Freeze (1987a, 1987b), Meyer and Brill (1988), Ahlfeld and Pinder (1988), and Meyer et al. (1989).

The simulation approach is computationally intensive. For example, in the given approach, in addition to the simulation of the hydraulic conductivity fields based on methods such as turning band method (Mantoglou, 1987), one also needs ground-water flow models [e.g., McDonald and Harbaugh (1988)] and mass transport models [e.g., Konikow and Bredehoeft (1984) and Goode and Konikow (1989)] in order to generate contaminant fields. These models are, in fact, the weakest links in the simulation approach. For instance, mass transport models have limited predictive skill in heterogeneous and anisotropic aquifers, i.e., those aquifers of greatest practical interest (Loaiciga 1988a). Furthermore, the mass transport phenomena are influenced by many chemical and biological processes that are still insufficiently understood and neglected in such models.

Despite these drawbacks, the simulation approach to ground-water quality monitoring network design offer an appealing flexibility to examine the efficiency of alternative network configurations and sampling frequencies, especially under relatively simple hydrogeologic settings. It also allows for the consideration of resource constraints, such as budgetary limits. Some degree of design optimization is possible, although computational requirements render it too cumbersome to consider but a limited number of network configurations.

Due to the linked nature of hydraulic conductivity and mass transport distribution generation, and the computational requirements, the simulation approach seems better tailored for problems involving contaminant monitoring at the field scale, as indicated by previous studies [e.g., Massmann and Freeze (1987a 1987b)]. It should be noted that the increased availability of cheap computational power and advances in mass transport modeling will most likely make the simulation approach a more attractive alternative for ground-water quality monitoring network design.

Variance-Based Approach

The second group of statistical techniques use additional statistical properties of the estimated values. More specifically, their basis is the estimation variance. As noted earlier, geostatistics offers estimation algorithms, where each estimate comes with an estimation variance. Some geostatisticians, such as Andre G. Journel (personal communication, 1989), view the estimation variance simply as a geometric indicator, due to the fact that it is primarily a function of the relative position of the estimated points and the measurement points. With this viewpoint, the estimation variance would be a questionable indicator of the accuracy of spatial estimation. There is, however, another point of view expressed by the majority of geostatistical hydrologists who argue that the estimation variance produced by kriging is a measure of the dispersion of the ensemble or collection of plausible values at the estimated point, and thus, may be regarded as a measure of the accuracy of the estimated value. Such a reasoning leads to the conclusion that the reduction in the estimation variance due to additional measurements may be regarded as an improvement in the accuracy, and therefore as an information gain. Thus in variance-based methods, the objective is to minimize the estimation variance or some function of it, subject to various constraints. Variance-based techniques have been used for the design of meteorological and hydrological networks by such authors as Fiering (1965), Matalas (1968), Bras and Rodriguez-Iturbe (1976a, 1976b), and Bastin et al. (1984), among others.

The estimation variance has some useful properties, including the fact that it does not depend on the individuals observation readings, which allow the planner to pose such question as "how much accuracy is gained if additional observations are made at location x or y." As noted by the ASCE ("Review" 1990b):

> . . . the worth of each sampling alternative can be evaluated before any new measurements are actually conducted. This is due to the fact that the uncertainty-reducing effectiveness of any sampling scheme depends only on the number and location of measurement sites, and not on the magnitude of measured values at those sites.

Among variance-based approaches are the following:

Global Method. The aim of this method is to identify the best pattern (e.g., square, triangular, or other geometric arrangement) and the best density (the number of points per unit area) of the sampling sites. The works of such authors as Olea (1984), Yfantis et al. (1987), and Christakos and Olea (1988) present some global (i.e., over the entire sampling domain) indices for the performance of a monitoring program, including the average or maximum variance of estimation. The choice of such global performance standards does not seem to be consistent with ground-water quality monitoring goals, where interest is commonly directed to more localized performance parameters, such as contaminant concentrations near a well or over a subregion of the study area. Nevertheless, the use of global performance criteria for ground-water quality monitoring network design can be justified for preliminary or exploratory layouts in regional-scale programs, since the analysis is expeditious and might provide a sound starting basis for more detailed posterior refinements of the monitoring network.

Variance-Reduction Analysis. This approach (Rouhani 1985) uses a methodical search for the number and locations of sampling sites that would minimize the variance of estimation error of the variable of interest, such as the concentration of a pollutant. The search for a ground-water monitoring network configuration starts with a number of existing sample wells to which additional wells from a pool of potential sites are added, one at a time. The site of each additional well is chosen to produce the largest reduction in the variance of estimation error. Sampling sites continue to be added until the estimation variance can no longer be (or can only marginally be) reduced, or when the marginal gain in statistical accuracy is outweighed by other constraints, such as limited budgets. If there are no originally existing wells, a subset of sampling locations must be selected based on hydrogeologic or some other relevant considerations. Thereupon, the monitoring network is expanded according to the primary objective of adding those wells that will contribute most to the reduction of the estimation variance.

Considering that the estimation variance does not depend on the measured values, it is possible to express (in closed form) the reduction in the estimation variance in terms of: (1) The variance obtained prior to the addition of the last sampling site; (2) the monitoring network's configuration; and (3) the mean and covariance of the environmental variable of interest (Rouhani, 1985). Therefore, the effect of adding one sampling site can be quantified without any difficulty and independently of the particular measurement value at that point.

The statistical nature of the variance-reduction approach limits its capability to incorporate complex hydrogeologic settings, and it is most useful when the environmental variable of interest has a homogeneous and isotropic spatial behavior. Previous applications ("Review" 1990b) indicate that the variance-reduction analysis is best suited for the determination of additional sampling sites, when the primary objective is to gain as much information as possible. In fact, the sequential nature of variance-reduction analysis appears to be consistent with the step-by-step nature of many ground-water quality monitoring activities.

Optimization Approach. In this approach, the ground-water quality monitoring network design is posed as a mathematical programming problem. Therefore, there is always an objective function, such as minimizing the estimation variance of ground-water quality properties (Knopman and Voss 1988; Loaiciga 1989) subject to different constraints, such as resource constraints; governing equations of physical processes (e.g., hydrodynamic dispersion); statistical constraints [e.g., accuracy of ground-water quality parameter estimates (Hsueh and Rajagopal 1988)]; and areal coverage of the monitoring network. The mathematical programming problem, represented by the optimization of the objective function, subject to the constraints, is then solved by appropriate algorithms.

In cases where the key outputs of the optimization approach are the location of sampling sites from a pool of potential sites, the corresponding programming problem usually requires the use of integer or binary variables. These variables reflect either the placement or the absence of an actual sampling site at a potential sampling location (Loaiciga 1988b; 1989a; Hsueh and Rajagopal 1988). The integer programming formulation has a long tradition in locational theory applications (Church and Revelle 1974; Meyer and Brill 1988). Examples in environmental monitoring are the early papers by Fiering (1965) and Darby et al. (1974). More general formulations, not limited to a finite number of predefined potential sites, that consider the entire continuum of points in the monitoring area as likely sampling locations have not as yet materialized in the hydrological literature.

The optimization approach is appealing because, in principle at least, it yields optimal sampling locations and sampling times while considering a variety of restrictions on the sampling plan. However, some of the most advanced applications reported to date (that incorporate the contaminant transport equations as constraints) have obvious limitations. Such limitations concern mainly the required simplifications of the hydrogeologic setting that must be conducted to prevent the network design problem from becoming cumbersome. Similarly, to other network design approaches that use ground-water mathematical models (e.g., simulation), the optimization approach offers a clear trade-off between the ability to model and optimize a network design problem on one hand, and the hydrogeologic simplifications that must be introduced on the other. Therefore, the optimization approach seems to be more promising as an analytical tool for regional ground-water quality monitoring network design, where the required hydrogeologic resolution is relatively coarse and easier to model.

A shortcoming of many optimization-based methods for ground-water quality monitoring network design is their static nature, i.e., they yield sampling networks that do not consider the iterative nature of many typical ground-water quality monitoring activities. In this instance, the optimized network must be considered as a preliminary layout to be subjected to subsequent modifications as new ground-water quality information is collected. Sequential procedures seem, at this point, the most likely new generation of optimization methods for ground-water quality network design. A step in this direction is the work by Graham and McLaughlin (1989a, 1989b) in which the monitoring network evolves as a function of newly sampled ground-water quality data.

For further applications of optimization techniques in network design in other related fields, such as hydrology, readers are referred to the papers by Carrera et al. (1984) and Hsu and Yeh (1989). It is worth noting that optimization models can be entirely deterministic, in which case they would fall in an entirely different category.

It should be noted that in all variance-based techniques (i.e., global, variance reduction, or optimization), it is implicitly assumed that the results of new measurement do not cause any significant change in the assumed statistical structure (e.g., covariance or semi-variogram) of the variable of interest. The assumption of the constancy of the statistical structure has been tested by Rouhani and Fiering (1986) for variance reduction analysis. They observed that "even slight levels of simulated noise in the input data cause significant changes in the general pattern of the estimated covariance function. On the contrary, the instability of the parameter space has a negligible effect on the action space (i.e., the results of sampling network design)." Therefore, it may be concluded that despite the questionable nature of the covariance constancy assumption, it does not necessarily influence the results significantly. Therefore, the variance-based techniques can generally be regarded as robust decision tools even when the given assumption cannot be fully maintained.

Finally, one of the major disadvantages of variance-based techniques stems from the fact that in a practical sense, the objective of minimizing the estimation variance is practically intangible and not easily understood by decision makers. For example, while a planner can easily evaluate the cost of an additional sampling, the resulting benefits (i.e., improvement in accuracy) remain rather elusive. In other words, minimization of the estimation variance, as desirable as it may be, is not necessarily a realistic goal in sampling design. As noted by the ASCE ("Review" 1990b), a more realistic approach is to minimize the negative impacts that inaccuracies have on the ultimate objective of the project. Rouhani (1985) proposes the use of loss functions in order to convert the increase in accuracy into a tangible monetary term. The derivation of a realistic loss function, however, is not an easy task, as is shown in such works as Bras and Rodriquez-Itrube (1976a, 1976b) and Bogardy and Bardossy (1985). The drawbacks of using the estimation variance as an objective function for ground-water quality network design are intimately related to the issue of using surrogate objectives (of which the estimation variance is an example) as proxies to ultimate objectives. As previously discussed, the latter objectives directly consider the value of ground-water quality data in making decisions that impact the environment or other tangible human activity.

Probability-Based Approach

In the presented variance-based approaches, the primary objective is to maximize the information gain, represented by the minimization of the estimation variance. However, as stated by Rouhani and Hall (1988), these approaches give more priority to points with high estimation variance, regardless of their estimated magnitudes. Such variance-based criteria are not suitable for a typical ground-water quality monitoring activity, where planners not only desire to gain as much information as possible, but also to be able to monitor areas where the variable of concern exhibits critical values. This implies that the selection criterion should be modified to include both accuracy (represented by the estimation variance) and the magnitude of the estimated values (such as contaminant concentrations). Rouhani and Hall (1988) proposed a network design problem that incorporates the level of the variable in question (e.g., contaminant concentration) and its variance of estimation. This was done by introducing the probability of exceeding a certain level of the field variable as the criterion to be controlled in the network design problem. The exceedance probability depends on the level and the estimation variance of subsurface contaminants, and, thus, these two parameters influence the selection of sampling sites. The method is flexible and by, for instance, reducing the exceedance probability level, Rouhani and Hall (1988) showed that their method would become similar to minimizing the estimation variance. They also showed that their method could identify critical sampling sites, i.e., those with a high likelihood of detecting high concentration levels. A case study of a probability-based approach as applied to the design of a regional ground-water quality monitoring in a shallow aquifer was presented by Rouhani and Hall (1988).

The work by Meyer and Brill (1988) introduced an iterative method that combines simulation and optimization to arrive at a network configuration that maximizes the probability of contaminant plume detection. In their approach, Meyer and Brill (1988) transform a problem of decision under uncertainty (i.e., selection the best sampling sites given the imperfect knowledge of contaminant distribution) into a deterministic optimization problem. This is done by first identifying the best set of plausible sampling sites via Monte Carlo simulation. Having identified those plausible sampling sites, an integer mathematical programming level was developed to select those sites that maximize the likelihood of detection of the occurrence of contaminants in the subsurface, while maintaining the size of the sampling network within acceptable limits.

Scheibe and Lettenmaier (1989) developed another probability-based approach to network design that relies on a hierarchical sampling design strategy. In their approach, there are three levels in the network design process: (1) A comprehensive contaminant, geographic, and hydrogeologic reconnaissance; (2) an estimation of the probability of contamination of wells; and (3) the selection of sampling sites (i.e., wells). Their method was applied to a regional-scale monitoring program involving agricultural pesticide contamination. The objective function of the network design method was to minimize the aggregate exposure risk of population centers served by water wells (the aggregate risk is defined as the sum of the product of contamination probability at a well times the population served by the well, where the sum is over all water wells in the study area).

We clarify that in this review paper, we reserve the appellative "risk-based" for those network design methodologies that consider exposure hazard and dose-response assessments in the characterization of risk posed by chemicals in ground water (*Risk* 1983). Otherwise, methods involving certain types of probabilistic analyses, such as the works by Rouhani and Hall (1988) and by Scheibe and Lettenmaier (1989), are termed "probability-based." The works by Massmann and Freeze (1987a), Anderson et al.

(1988), and Reichard and Evans (1989) are partial attempts to pose ground-water quality network design within a risk-based framework.

SUMMARY AND CONSLUSIONS

Our review has identified the most prominent approaches to ground-water quality monitoring network design. These approaches serve as a guideline for the preliminary analysis of network designs. Quite often, the final and implementable sampling program is of a dynamic nature and influenced by institutional factors. In addition, considerations relevant to quality control and quality assurance (QC/QA) of a monitoring program (Olson and Heffner, 1989) and statistical data analysis (Gilbert 1987; McNichols and Davis 1988; *Statistical* 1989) play an important role in ground-water quality monitoring. Quality control and quality assurance considerations might impose restrictions on the (temporal) sampling frequency, the number of replicate ground-water samples to be collected, and the cost of water sampling collection. Statistical considerations (that could be introduced by QC/QA protocols) might determine the minimum number of ground-water samples needed in the collection process and the geometric distribution of sampling sites (Loaiciga and Hudak 1989).

In summary, most of the existing ground-water quality network design methods make several important simplifications. First, the majority of these methods assume that decisions to monitor are made once and for all, without the opportunity for feedback and adjustment. Second, most network design methods often use surrogate objectives for cost and health criteria. Third, in many instances, those methods oversimplify the hydrogeologic environment, with the methods' usefulness in more realistic settings remaining unproven. Fourth, the presented methods generally do not consider geophysical and other inexpensive vadose-zone monitoring methods for obtaining information that has been shown to be helpful in many circumstances (Russell and Higer 1988). For example, electrical conductivity measurements or soil tests can be used to roughly delineate contaminant plume at a low cost; their results can then be used as a basis for more efficient network designs. In some instances, it is possible to model the detection probabilities of geophysical methods and to include these in an evaluation of alternative monitoring methods (Greenhouse and Monier-Williams 1985). Table 2 summarizes key features of the described approaches to ground-water quality monitoring design. The Xs marked under each of the columns in Table 2 indicate which features apply to which approaches. A question mark appended to an X signals that the subject feature is only weakly applicable to the corresponding approach. Table 2 provides a general guide to the potential strengths and obvious weaknesses of the various approaches to ground-water quality network design. The process of selecting a method for network design must also consider: (1) The overall purpose of monitoring (e.g., ambient monitoring or compliance monitoring); (2) the type of criteria to be satisfied (i.e., ultimate objectives, directly related to the use of data in decision making, or surrogate objectives, such as statistical criteria weakly related to the value of information in decision making); and (3) the dynamic nature of ground-water quality and its interaction with decision-making processes that are influenced by ground-water quality data.

In spite of the complexities associated with ground-water quality monitoring, it is fair to state that over the last two decades there have been significant developments that now permit the application of methodical and

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TABLE 2. Key Features of Approaches to Ground-Water Quality Monitoring Network Design

	APPROACH					
		STATISTICAL				
			Variance-Based			
Features (1)	Hydrogeological (2)	Simulation (3)	Global (4)	Variance-reduction (5)	Optimization (6)	Probability-based (7)
Scale:						
Field	x	x		x?	x?	x
Regional			x	x	x	x
Constraints:						
Resource	x	x	x	x	x	x
Statistical		x	x	x	x	x
Hydrogeological	x	x			x	x?
Data intensive	x	x				
Complex hydrogeology:	x	x?				
Temporal effects						
Sampling Frequency	x?	x			x?	
Representative reference(s):	Everett (1980); *RCRA* (1986)	Massman and Freeze (1987a, 1987b); Meyer and Brill (1988)	Olea (1984)	Rouhani (1985)	Hsueh and Rajagopal (1988); Loaiciqa (1989)	Rouhani and Hall (1988); Scheibe and Lettenmaier (1989)

testable approaches to ground-water quality monitoring network design. Improvements in the modelling of mass transport and ground-water flow, as well as in field instrumentation and hardware/software, have proven to be highly beneficial in the development of standardized methods and well-defined approaches for ground-water quality network design. Equally helpful have been the advances in risk assessment and geostatistical analyses. Currently, we have a number of network design methods applicable to a variety of ground-water quality monitoring situations. The state of the art is that they mostly serve as a tool for preliminary analysis and design. The opinion of the Task Committee on Ground Water Quality Monitoring is that as there continues to be advances in hydrogeochemistry, ground-water hydrology, and risk and geostatistical analyses, methods for ground-water quality monitoring network design will be improved and refined, and they will become ever more useful in the important mission of environmental protection.

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