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A Location Modeling Approach for Groundwater Monitoring Network Augmentation

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A heuristic approach based on facility location theory effectively augments groundwater quality monitoring configurations through the strategic siting of additional wells. The approach is an alternative to variance-based schemes previously established for the augmentation problem. The method developed herein is practical in that it (1) is relatively easily understood, (2) is not difficult to implement and solve, and (3) does not require a large number of preexisting observation points. The approach has been applied to a case study involving a landfill-contaminated buried valley aquifer in southwest Ohio. Configurations derived by the heuristic approach exhibit two key characteristics: (1) location of wells near areas of high estimated contaminant concentration and (2) interwell separation distances that facilitate areal plume coverage.

INTRODUCTION

Groundwater quality monitoring network design is an expanding field in an early stage of development. An important application in this field is the determination of a configuration of well sites that optimizes the probability of detecting contaminant from a waste impoundment, such as a solid waste landfill. Results of previous studies that have focused on this application attest to the potential utility of network design methodology for contaminant detection from new or proposed waste facilities. However, many existing landfills, especially those constructed decades ago (e.g., in the 1970's or earlier), have already contaminated underlying aquifers. Contamination from older facilities is an inevitable consequence of insufficient or inadequately constructed containment structures. For an older facility, where an existing monitoring configuration is composed of a limited number of inadequately distributed wells, a location methodology may be used to augment the existing network. The objective of monitoring network augmentation is to locate additional wells to gain further information on maximum contaminant concentrations and the spatial extent of the contaminant field. This information can be used to guide future decisions involving, for example, the locations of extraction wells for aquifer remediation and water supply well abandonment. This paper presents a heuristic approach for augmenting an existing network in an aquifer contaminated by an older landfill. The approach, based on facility location theory, is an alternative to statistically based variance reduction approaches that have been established for the augmentation problem.

BACKGROUND

At the site-specific or local scale, approaches for locating groundwater quality monitoring wells can be classified as either network design or network augmentation. Network design involves the determination of a configuration of monitoring wells for a site not characterized by existing wells. Without existing wells there is no information regarding the presence of contaminants in the groundwater environment, nor is there adequate information on aquifer properties. Typically, parameter uncertainty is taken into account using Monte Carlo simulations. In the simulation approach the hydraulic conductivity of a porous medium is considered a random field. Because hydraulic conductivity exerts an important control on groundwater velocity distribution and therefore on the advection and dispersion of contaminants, both velocity and pollution distributions are stochastic or random fields as well. By generating multiple synthetic distributions of hydraulic conductivity, for each of which there is a corresponding contaminant distribution, it is possible to assess the performance of a given monitoring network. Applications employing these techniques typically involve the siting of a fixed number of wells in an arrangement that maximizes the probability of release detection from a new or proposed facility [e.g., *Massmann and Freeze*, 1987*a*, *b*; *Meyer and Brill*, 1988].

Network augmentation is a distinctly different problem. In the augmentation problem, monitoring wells are added to a preexisting network for the purpose of contaminant plume characterization. Information on aquifer properties and contaminant concentrations obtained from preexisting wells, and the locations of these wells, are considered in the augmentation process. Variance-based approaches [e.g., *Rouhani*, 1985; *Loaiciga*, 1989; *Graham and McLaughlin*, 1989*a*, *b*] have been established for the problem of ground water quality monitoring network augmentation in local-scale settings. The variance-reduction approach involves a methodical search for the number and locations of sampling sites that minimize the variance of estimation error of pollutant concentration [*Bastin et al.*, 1984; *Rouhani*, 1985; *Task Committee on Geostatistical Techniques in Geohydrology*, 1990*a*, *b*]. The approach is characterized by some important limitations.

1. The structural analysis, which quantifies spatial correlation throughout the model domain [e.g., *Journel and Huijbregts*, 1978), is derived solely on the basis of information obtained from preexisting sampling sites. In many cases a large number of initial sites may be unavailable. It is assumed that the structural analysis remains invariant under the addition of new sampling sites.

2. In many variance-based approaches the objective

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gives more priority to points with high estimation variance, regardless of estimated magnitude [*Loaiciga et al.*, 1992]. This result is not desirable from an applied standpoint, where it may be important to locate wells near areas of high contaminant concentration, particularly in the internal portions of a plume where preexisting coverage may be insufficient.

3. Many applications of the approach seek to minimize the variance of contaminant concentration estimation error within some finite estimation region [e.g., *Loaiciga*, 1989]. In these applications the variance-reduction technique may not generate a separation of well sites to facilitate areal coverage over an entire plume. On the contrary, wells are clustered near the estimation region.

4. Finally, the statistical nature of the 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 [*Loaiciga et al.*, 1992].

The methodology developed herein provides an alternative to the statistically based variance-reduction approach for groundwater quality monitoring network augmentation. The heuristic approach employs facility location and numerical mass transport models and is not characterized by some of the limitations that restrict the applicability of variance-based approaches.

HEURISTIC DEVELOPMENT

The initial step in the hueristic approach is the discretization of a model domain into a network of potential sampling sites (nodes). Each node is assigned a weight quantifying its relative importance as a sampling site. Weights are set equal to above-background contaminant concentrations calculated by a numerical mass transport model. The geometry of the grid of potential sites is dependent, in part, on the type of mass transport model employed. In applications using finite difference models, grid orthogonality is required. Spacing between nodes is based on several criteria, including the contaminant transport properties of the aquifer, the geometry of the study area, spacing between existing wells, and computational considerations. The roles of each of these criteria in defining the grid are discussed in a subsequent section, with reference to a case study application. Given a grid of weighted nodes and the locations of existing well sites (assigned to nodes closest to actual location), a modified version of the maximal covering location model [*Church and Revelle*, 1974] locates a specified number of additional wells. The model formulation is presented below.

$$\text{Max } Z = \sum_{i \in I} W_i y_i \tag{1}$$

subject to

$$\sum_{j \in N_i} x_j \geq y_i \qquad \text{for each } i \in I \tag{2}$$

$$\sum_{j \in J} x_j = P \tag{3}$$

$$x_j = (0,\ 1) \qquad \text{for all } j \in J \tag{4}$$

$$x_j = 1 \qquad \text{for all } j \in J_p \tag{5}$$

$$y_i = (0,\ 1) \qquad \text{for all } i \in I \tag{6}$$

where

W_i weight at node i;
I set of nodes in discretized network;
J set of possible well sites;
J_p set of nodes J occupied by preexisting wells;
$N_i = [j | d_{ij} \leq S]$;
d_{ij} shortest distance from node i to node j;
S covering distance threshold;
$x_j = 1$ if a well is installed at site j; 0 otherwise;
$y_i = 1$ if node i is covered; 0 otherwise;
P total number of wells (preexisting and added) to be located.

The objective (1) is to maximize coverage, where coverage is a function of distance. A well sited on a node covers (contributes the weight values of) the corresponding node and, depending on the value of S, may cover nearby nodes. A node is covered if at least one well is located within a distance S of the node, as specified by constraint (2). Constraint (3) requires that P total (preexisting and added) wells be located by the model. Contraints (4) and (6) impose binary restrictions on model variables, and constraint (5) requires that preexisting wells are included in the model solution.

The location model is used to generate alternate configurations for a specified number of wells. Each configuration corresponds to the solution for a single value of S (explained below), and configurations are generated until a specified areal plume coverage is achieved. Plume coverage is defined as the percent of nodes with weight values above zero that are covered by one or more wells. Decision variables include the number of wells to be added, the set of existing well sites to be retained in the expanded network (may be a subset of the total set of existing sites), and an areal plume coverage goal. Budgetary and/or regulatory restrictions may influence the choice of values for decision variables.

The parameter S controls the pattern of well siting. For the purpose of maximizing the model objective function it is generally advantageous to avoid coverage overlap. This condition occurs when a node is covered by more than one well. Standard separation represents the minimum distance between wells needed to ensure a condition of no coverage overlap. For a given formulation the standard separation distance is directly related to the value for S. Larger standard separation distances, and a related tendency for greater interwell separation in model solutions, are achieved by increasing the value of S. Solutions are obtained for progressively higher values of S, until a specified plume coverage is achieved. The succession of values taken on by S (S_1, S_2, S_3, $\cdots$) is determined by the geometry of the grid. Values for S are incremented such that there is a minimum possible increase in the number of nodes that can be covered by a single well from one S value to the next, as follows. For a given grid an arbitrary reference node is identified. The minimum S value, S_1, is set equal to zero. S_2 is equal to the distance from the reference node to the nearest neighboring node(s), S_3 is the distance from the reference node to the nearest neighboring nodes that are beyond a distance S_2, S_4 is the distance to the nearest nodes beyond a distance S_3, and so on. The parameter S is not directly related to the mass transport properties of the aquifer. These properties are taken into consideration in the design of the grid of possible well sites. Variation in the value of S constitutes a mechanism to influence the pattern of sited wells until a specified minimum coverage is obtained.

There are some exceptions to the tendency for standard well separation. A condition where wells are spaced at distances within the standard separation is referred to as well clustering. There are two specific situations that might lead to a condition of well clustering.

1. Near the contaminant source, for example, a relatively close spacing of preexisting wells may not permit the placement of an intermediate additional well without a coverage overlap. However, despite the penalty incurred by overlapping coverage (the weights of overlapped nodes are counted only once), the maximum contribution to the objective function may still be achieved by siting the added well in the intermediate position. This occurs when the weight values for the uncovered intermediate nodes are relatively high and cannot be covered with an alternative placement of the additional well. Stated differently, it may be better to cover fewer nodes with high weight values, near the source, than to locate the added well further away from the source and cover more nodes without coverage overlap.

2. In an aquifer region where a group of highly weighted nodes are bordered by nodes with significantly lower weight values (i.e., where sharp concentration gradients exist), it may be most beneficial, for the purpose of optimizing the objective function to locate added wells such that a maximum number of higher weighted nodes are covered, at the expense of possible coverage overlap.

Effectively, exceptions 1 and 2 allow well clustering, particularly near the contaminant source. The general tendency for standard separation results in greater uniformity in areal coverage in regions more distant from the source. The tendency for increased well separation with distance from the source is consistent with the general tendency for a plume to spread with downgradient migration in response to hydrodynamic dispersion. Greater well densities are warranted near the margin of the surface impoundment, where the emanating plume is at its narrowest. The finer spacing facilitates detection of maximum concentrations in the central portion of the plume. With increased distance the areal extent of the plume is better characterized by a configuration of wells spaced further apart. The tendency for standard well separation and examples of exceptions 1 and 2 are illustrated with reference to an application in a subsequent section.

Weights at individual nodes in the discretized network are set equal to contaminant concentrations calculated by a numerical mass transport model calibrated to concentrations measured at existing wells. The numerical model-calculated distribution that best matches observed concentrations at corresponding locations is used for assigning nodal weights. The model-calculated distribution is an approximation of the unknown distribution of contaminant. This approximation is needed to quantify the importance of sampling at potential sites. Intuitively, the higher the (estimated) contaminant level, the more important it is to monitor at the corresponding site. There is also importance in monitoring throughout an area potentially affected by migrating contaminant, including plume boundaries. This property is accomplished by interwell separation, which is driven by the parameter S, and areal plume coverage specification. For a given application the uncertainty in numerical model generated concentrations

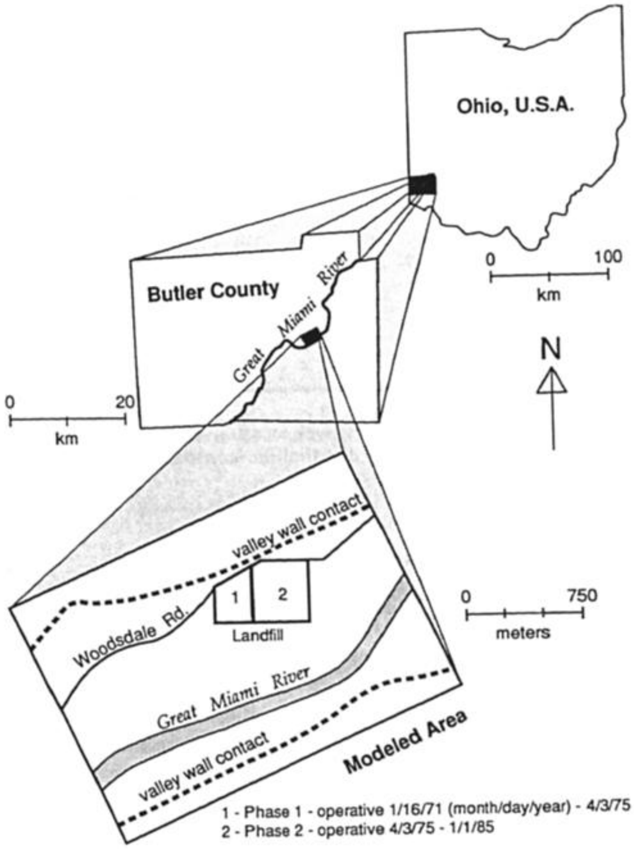


Fig. 1. Location of study area.

is reflected in differences between observed and model-calculated values at corresponding locations (i.e., at preexisting monitoring wells). Clearly, the unknown concentration distribution may be approximated in other ways (e.g., geostatistical techniques). However, as suggested previously, an adequate structural analysis cannot be performed with a sparse number of preexisting sites, which is typical for most augmentation problems. Furthermore, by utilizing a numerical model calibration procedure, the concentrations measured at existing wells are included in the analysis, as are the groundwater flow and mass transport properties of the aquifer.

In summary, the sequence of steps involved in implementing the heuristic approach are (1) definition of a grid containing possible monitoring sites, (2) location of existing well sites on the grid, (3) groundwater flow and mass transport model calibration to concentrations obtained from existing wells, (4) input of calibrated concentrations as nodal weights in the modified maximal covering location problem formulation, and (5) solution of the formulation for progressively higher values of S until a specified areal coverage is achieved.

APPLICATION

The network augmentation approach was applied to a landfill-contaminated buried valley aquifer in southwest Ohio. The solid waste landfill, located in east central Butler County on the flood plain of the Great Miami River, received domestic and commercial waste from 1971 to 1985 in two phases of operation (Figure 1).

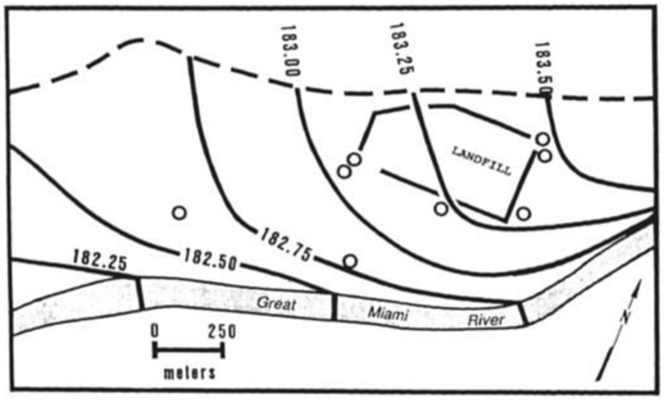


Fig. 2. Existing monitoring well locations (circles) and observed steady state hydraulic head distribution (contours in meters above mean sea level).

Hydrogeologic Setting

The buried valley aquifer consists of glaciofluvial deposits comprised predominantly of coarse sand and gravel. Silt and clay deposits are present in thin, laterally discontinuous lenses. Results of field tests conducted by *Hudak and Loaiciga* [1991] indicate that the aquifer is characterized by an average hydraulic conductivity of approximately 10^{-3} m/s. Unconsolidated deposits range from 6 to 12 m in thickness and overlie shale bedrock. The regions to the immediate northwest and southeast of the study area (Figure 1) are upland surfaces underlain by glacial till. Till deposits range from 1 to 12 m in thickness and directly overlie the shale bedrock [*Spieker*, 1968]. The shale bedrock and glacial till are characterized by low hydraulic conductivities (estimated less than 10^{-8} m/s). Approximate boundaries between the upland surfaces and the unconfined aquifer are represented by the valley wall contacts depicted in Figure 1. No geologic boundaries are present transverse to the axis of the buried valley at the northeast and southwest margins of the study area. Topography is uniformly flat, with an average elevation of about 185 m above mean sea level. Depth to groundwater characteristically ranges from 3 to 5 m. The river is effluent (gaining) on a year-round basis and in hydraulic connection with groundwater in the underlying aquifer (Figure 2).

Grid Definition

The discrete network of candidate nodes and locations of existing well sites is illustrated in Figure 3*a*. Nodes occupy the region north of the river. In designing the network, the following properties were sought:

1. An orthogonal network was used, corresponding in geometry to the finite difference grid employed with the associated numerical model.
2. An overall grid was designed that conformed to the general shape of the study area (i.e., with one coordinate direction approximately parallel to the natural hydrogeologic boundaries). This type of grid is most efficient in covering the region of interest.
3. Interwell spacing was a minimum of 50 m to effectively cover a large extent of the model area. This value corresponds to the approximate minimum distance between wells in the existing network.
4. The entire region north of the river was covered uniformly with candidate nodes. Areas throughout this region could potentially be affected by migrating contaminant.
5. No more than a few hundred potential well sites were selected. This property is necessary to limit computational requirements.

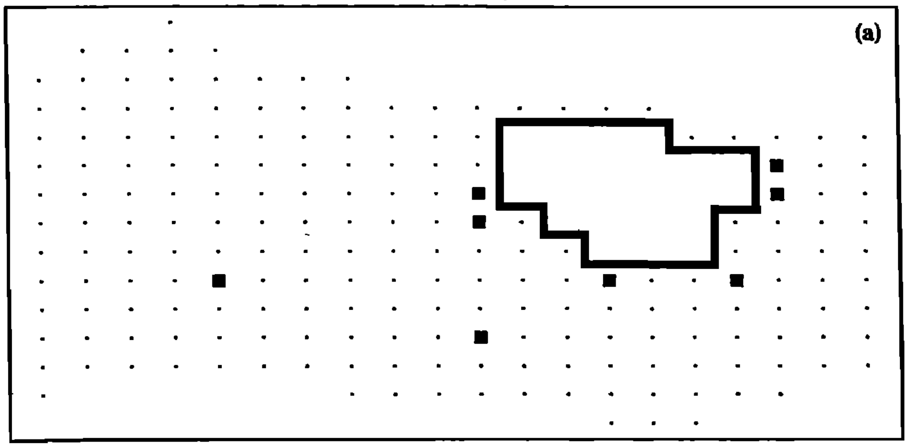
The exact nature of the grid employed for a monitoring network augmentation problem is somewhat arbitrary, but the properties listed above constrained the grid appropriate for the present application to resemble that shown in Figure 3*a*. Node spacing is 250 ft (76.2 m) along columns and 375 ft (114.3 m) along rows. For a given application, grid definition is subject to the mass transport properties of the aquifer under study. In general, heterogeneous conditions and/or high groundwater velocities lead to the need for smaller internode separation distances. Where preferential migration pathways along zones of relatively high hydraulic conductivity can be defined, grid spacing should not exceed the widths of such zones, or else contaminant in these areas could go uncharacterized. An important limitation of grid-based approaches is the restriction of well locations to a predefined set of possible sites. Computational limitations dictate minimum spacings between potential sites of the order of at least tens of meters in most cases. Where spatial variations in concentration are large near potential well sites, grid-based schemes such as the approach developed herein may provide insufficient resolution.

Nodal Weights

A groundwater flow and mass transport model (method of characteristics model) [*Konikow and Bredehoeft*, 1988] was calibrated to hydraulic head and contaminant concentration levels measured at preexisting well sites (Figure 2). The wells sample groundwater from intermediate depths within the saturated zone (i.e., within the 3–7 m depth interval below ground surface). *Hudak and Loaiciga* [1991] provide a detailed description of field data collection procedures and calibration results. The numerical grid corresponds to the grid depicted in Figure 3*a*. Chloride was used as a tracer in modeling contaminant migration. Numerical model calibration resulted in the concentration distribution illustrated in Figure 3*b*. Contaminant concentrations generated by the numerical model in the optimal calibration run (Figure 3*b*) were input as the weights for candidate well sites in the monitoring network augmentation heuristic.

Model Solution

The location model represented by equations (1)–(6) was solved with the LINDO (Linear, Interactive, and Discrete Optimizer) mathematical programming package [*Schrage*, 1987] on an IBM RT running AIX (the IBM version of the UNIX operating system). Most model formulations were solved in 3–5 min and, in the majority of cases, the relaxed linear programming solutions were all integer (no branch-and-bound was necessary). The coverage model was solved for the siting of 10 additional wells to the preexisting eight-well network. The decision variable P is thus equal to 18 for this application. An areal coverage goal of two thirds of the number of contaminated nodes (i.e., nodes with numerical model-estimated concentrations above background) was specified. The succession of S values used is illustrated in Figure 4. Values are given in grid units, where



(b)

+	+	+	1	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
+	2	3	3	4	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
2	3	4	5	7	8	12	12	+	+	+	+	+	+	+	+	+	+	+	+
4	5	6	7	10	14	17	22	28	36	45	50	53	66	74	+	+	+	+	+
5	7	9	11	14	21	31	39	48	67	140	*	*	*	*	50	24	7	1	0
6	8	11	14	18	26	37	52	76	110	166	*	*	*	*	*	*	18	3	1
7	9	12	15	21	30	45	60	92	126	181	*	*	*	*	*	*	12	2	0
7	10	13	17	25	38	50	66	98	126	169	189	*	*	*	*	68	5	0	0
7	9	14	19	27	39	53	69	98	123	146	168	187	*	*	*	29	1	0	0
6	8	14	21	28	39	52	67	93	117	129	147	163	167	147	118	19	0	0	0
4	7	12	19	27	36	50	65	83	104	115	126	142	133	107	74	8	0	0	0
3	5	7	17	23	35	49	61	79	92	105	112	126	113	89	51	6	0	0	0
3	5	7	16	21	30	41	55	71	81	92	106	108	92	70	45	4	0	0	0
2	+	+	+	+	+	+	54	62	69	80	92	85	77	57	25	3	0	+	+
+	+	+	+	+	+	+	+	+	+	+	+	+	58	43	24	+	+	+	+

Fig. 3. (a) Discretized field of possible monitoring sites and existing wells (squares). (b) Chloride concentration distribution generated by numerical mass transport model; numbers correspond to concentrations above background, in parts per million, calculated at centers of numerical model cells; plus, inactive cell; asterisk, contaminant source node; scale, 375 ft (114.3 m) between nodes along rows.

one unit is equal to the distance between adjacent nodes along rows. Note that the values are chosen giving consideration to the exact nature of the grid employed. Each successive S value results in the minimum increase in covering potential from the previous value.

The solution for $S = 0$ is illustrated in Figure 5*a*. Under this condition, sited wells have no covering potential. Wells contribute only the weight values of the corresponding nodes. There is no spatial interaction between sited wells, the location of a well at a node does not influence the possible location of a well at a nearby node. The resulting distribution is characterized by a clustering around high (estimated) concentration areas downgradient of the source, but only 18 (10%) of 181 contaminated nodes are covered. An increase in the value of S to 0.67 generated the configuration illustrated in Figure 5*b*. This configuration covers 49 (27%) of 181 contaminated nodes and is characterized by a greater degree of interwell separation than the previous solution. Solutions for S values of 1.00, 1.20, and 1.33 are illustrated in Figures 5*c*, 5*d*, and 6*a*, respectively. These configurations cover 41, 64, and 69% of contaminated nodes. The desired areal plume coverage (67%) is obtained with the solution for $S = 1.33$ (Figure 6*a*). The solution in Figure 6*a* is retained as the optimal configuration.

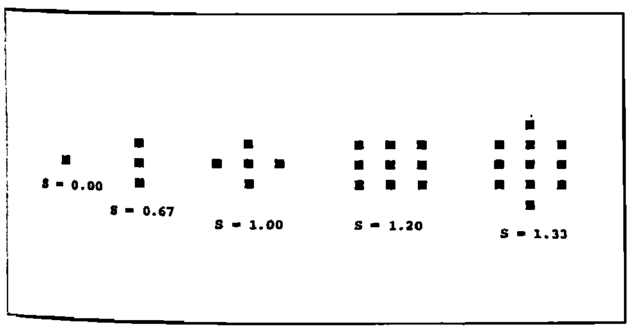


Fig. 4. Coverage patterns for different values of S; squares correspond to nodes that can be covered by the central node.

Alternate Weighting Schemes

Weights obtained by numerical model calibration can be modified to reflect the emphasis of a particular monitoring program. For example, where a contaminant plume is in the vicinity of water supply wells, nodes can be assigned weights

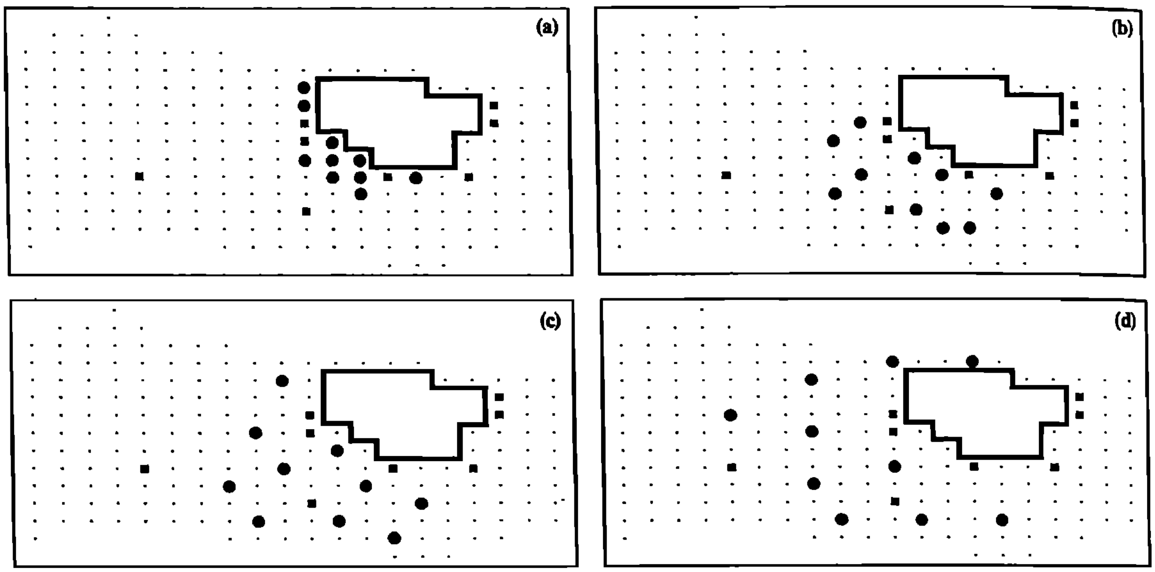


Fig. 5. Solutions for S = (*a*) 0.00, (*b*) 0.67, (*c*) 1.00, and (*d*) 1.20; dots represent added wells.

of the form C/D_s, where C is calibrated contaminant concentration, as before, and D_s, is the distance between a node and the nearest supply well. In the case study discussed above, six small-capacity domestic supply wells are located along the western downgradient margin of the study area. The optimal heuristic solution for weights of the form C/D_s, with a two-thirds areal coverage specification, as before, is illustrated in Figure 6*b*. Both optimal heuristic solutions (Figures 6*a*–6*b*) emphasize the aquifer region downgradient of the contaminant source for additional monitoring. However, the solution in Figure 6*b* gives greater emphasis to the downgradient area close to the water supply wells. The results outlined in Figures 6*a*–6*b* indicate that the heuristic solution is sensitive to choice of nodal weights. For the weighting schemes outlined above the choice of units for a particular parameter (e.g., concentration C) do not affect the heuristic solution. Where a unit conversion is obtained by multiplying by a constant, the relative (percent) difference between nodal weights is unchanged.

Timing of Well Siting

An optimal configuration for a given combination of S and P cannot be obtained by retaining a previous configuration resulting from the identical S value and a lower P value. For example, the configuration for S = 1.33, P = 18 (i.e., 10 added wells) illustrated in Figure 6*a* cannot be achieved by two five-well sitings. The solutions for two five-well sitings are illustrated in Figures 6*c*–6*d*. In Figure 6*c*, five-wells are added to the preexisting eight-well network. The 13 well sites in Figure 6*c* are then retained as preexisting sites, and five wells are added, resulting in the configuration illustrated in Figure 6*d*. The configuration in Figure 6*d* differs from the one-time 10-well siting in Figure 6*a*. In particular, there is a

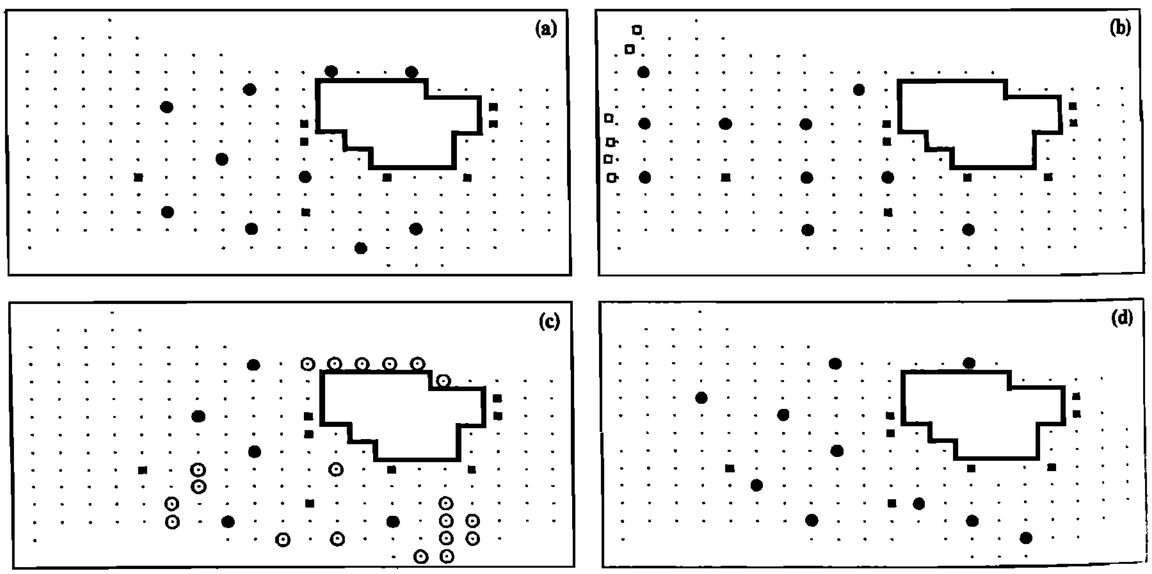


Fig. 6. (*a*) Optimal solution (S = 1.33). (*b*) Optimal solution for weights of the form C/D_s (S = 1.20); open squares designate supply well sites. (*c*) Solution for S = 1.33 with five wells added to an eight-well network (weights of the original form C); open circles designate uncovered nodes. (*d*) Solution for S = 1.33 with five wells added to the 13-well network in Figure 6*c*.

greater degree of clustering in the solution in Figure 6*d*, suggesting inefficient covering. The inefficiency is further suggested by a lower objective function value (8418) compared to that obtained for the solution in Figure 6*a* (8549). With regard to the solution resulting from two five-well sitings, note that the configuration obtained after the first set of five-wells are added is characterized by a number of uncovered contaminated nodes (Figure 6*c*). These nodes have high (estimated) concentration values and would contribute significantly to the value of the objective function if covered. Many of the second group of five added wells are located to cover these nodes. However, because the first five sites are retained, there is a significant degree of coverage overlap. By siting 10 additional wells at once, a more efficient coverage can be obtained, characterized by a low degree of coverage overlap and a small number of uncovered sites in the internal part of the contaminated area (Figure 6*a*). Therefore with regard to the timing of well siting it is inefficient to derive a final configuration by retaining previous configurations with fewer wells.

CONCLUSIONS

A heuristic approach has been developed for upgrading existing monitoring well configurations in field-scale groundwater contamination settings. The method is most applicable to situations where (1) older waste facilities have contaminated underlying aquifers and (2) leachate plumes are partially characterized by a relatively sparse, preexisting network. The approach developed herein constitutes a viable alternative to variance-based approaches in groundwater quality monitoring network augmentation. A large number of preexisting sampling sites is not as critical as in many variance-based applications. Furthermore, the approach is not difficult to implement and results in intuitively logical groundwater monitoring well configurations. The approach sites wells in a strategic manner. Additional wells monitor maximum concentrations of contaminant in the aquifer while maintaining well separation for areal characterization of the contaminant plume. Nodal weights, derived from a numerical model calibrated to concentrations measured at existing monitoring wells, quantify the relative importance of sampling throughout the model domain. The use of numerical model calibration, a procedure commonly employed in engineering practice, may facilitate the practical applicability of the approach.

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