REGIONAL SCALE GROUND WATER
QUALITY MONITORING:
METHODS AND CASE STUDIES

by

Paul F. Hudak
Hugo A. Loaiciga
Miguel A. Mariño

University of California
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REGIONAL SCALE GROUND WATER QUALITY MONITORING:

METHODS AND CASE STUDIES

by

Paul F. Hudak
Department of Geography
University of North Texas

Hugo A. Loaiciga
Department of Geography and
Environmental Studies Program
University of California, Santa Barbara

and

Miguel A. Mariño
Department of Land, Air, and Water Resources and
Department of Civil Engineering
University of California, Davis
ABSTRACT

Mathematical programming models developed herein locate monitoring wells in uncontaminated, multi-layered ground water flow systems at risk of widespread pollution from overlying waste facilities. Potential well sites are assigned weights quantifying monitoring value, which is assessed on the basis of detection and exposure criteria. Detection criteria include the location of a site with respect to the source of contamination and potential contaminant plumes, and ground water velocity. Exposure criteria, used to assess exposure hazard at water supply wells, include user population and distances between a supply well and probable areas of contamination. The derived weights are integrated into binary integer mathematical programming problems. Formulated problems, based on facility location theory, can be solved using the branch-and-bound technique.

Model-derived and previously established monitoring networks were tested on the basis of plume detection and characterization criteria. On a 100-point rating scale developed to quantify composite detection and characterization efficiency, network design models scored eighty-five and seventy-eight, compared to a score of seventy-five for pre-existing monitoring networks. The most comprehensive model sites wells relatively close together near the source of contamination, facilitating early contaminant release detection, and further apart downgradient, resulting in areal coverage for plume characterization. The established strength of the developed models in the fundamental properties of detection and characterization attests to the utility of three-dimensional binary integer programming models for ground water quality monitoring network design at the regional scale.

Key Words: Decision Models, Ground Water Quality, Mathematical Models, Optimization.
# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Pages</th>
</tr>
</thead>
<tbody>
<tr>
<td>LIST OF TABLES</td>
<td>v</td>
</tr>
<tr>
<td>LIST OF FIGURES</td>
<td>vi</td>
</tr>
<tr>
<td>NOTATION</td>
<td>x</td>
</tr>
<tr>
<td>1. INTRODUCTION</td>
<td>1</td>
</tr>
<tr>
<td>2. LITERATURE REVIEW</td>
<td>2</td>
</tr>
<tr>
<td>2.1 Hydrogeologic Approach</td>
<td>2</td>
</tr>
<tr>
<td>2.2 Simulation Approach</td>
<td>2</td>
</tr>
<tr>
<td>2.3 Variance-Based Approaches</td>
<td>3</td>
</tr>
<tr>
<td>2.4 Optimization Approach</td>
<td>4</td>
</tr>
<tr>
<td>2.5 Summary</td>
<td>4</td>
</tr>
<tr>
<td>3. MODEL THEORY AND DEVELOPMENT</td>
<td>5</td>
</tr>
<tr>
<td>3.1 Basic Model</td>
<td>7</td>
</tr>
<tr>
<td>3.2 Modified Weighted Benefit Coverage Model (MWB Model)</td>
<td>11</td>
</tr>
<tr>
<td>3.3 Nodal Weights</td>
<td>17</td>
</tr>
<tr>
<td>3.4 Upgradient Monitoring</td>
<td>21</td>
</tr>
<tr>
<td>3.5 Temporal Changes in Monitoring Configuration</td>
<td>21</td>
</tr>
<tr>
<td>4. CASE STUDY 1</td>
<td>23</td>
</tr>
<tr>
<td>4.1 Site Description and Background</td>
<td>23</td>
</tr>
<tr>
<td>4.2 Hydrogeology</td>
<td>23</td>
</tr>
<tr>
<td>4.3 Model Boundaries and Discretization</td>
<td>23</td>
</tr>
<tr>
<td>4.4 Numerical Modeling</td>
<td>26</td>
</tr>
<tr>
<td>4.5 Network Configurations</td>
<td>27</td>
</tr>
<tr>
<td>4.5.1 Original Network</td>
<td>27</td>
</tr>
<tr>
<td>4.5.2 Basic Model Network</td>
<td>27</td>
</tr>
<tr>
<td>4.5.3 MWB Model Networks</td>
<td>28</td>
</tr>
<tr>
<td>4.6 Model Testing</td>
<td>28</td>
</tr>
<tr>
<td>4.6.1 Detection Efficiency</td>
<td>28</td>
</tr>
<tr>
<td>4.6.2 Characterization Efficiency</td>
<td>29</td>
</tr>
<tr>
<td>5. CASE STUDY 2</td>
<td>38</td>
</tr>
<tr>
<td>5.1 Site Description and Background</td>
<td>38</td>
</tr>
<tr>
<td>5.2 Hydrogeology</td>
<td>38</td>
</tr>
<tr>
<td>5.3 Model Boundaries and Discretization</td>
<td>39</td>
</tr>
<tr>
<td>5.4 Numerical Modeling</td>
<td>42</td>
</tr>
<tr>
<td>5.5 Network Configurations</td>
<td>43</td>
</tr>
<tr>
<td>5.5.1 Layer 1</td>
<td>43</td>
</tr>
<tr>
<td>5.5.1.1 Original Network</td>
<td>43</td>
</tr>
<tr>
<td>5.5.1.2 Basic Model Network</td>
<td>43</td>
</tr>
<tr>
<td>5.5.1.3 MWB Model Network</td>
<td>44</td>
</tr>
<tr>
<td>5.5.2 Layer 2</td>
<td>44</td>
</tr>
<tr>
<td>5.5.2.1 Original Network</td>
<td>44</td>
</tr>
<tr>
<td>5.5.2.2 Basic Model Network</td>
<td>44</td>
</tr>
<tr>
<td>5.5.2.3 MWB Model Network</td>
<td>44</td>
</tr>
<tr>
<td>5.5.3 Layer 3</td>
<td>45</td>
</tr>
<tr>
<td>5.5.3.1 Original Network</td>
<td>45</td>
</tr>
<tr>
<td>5.5.3.2 Basic Model Network</td>
<td>45</td>
</tr>
<tr>
<td>5.5.3.3 MWB Model Networks</td>
<td>45</td>
</tr>
<tr>
<td>5.6 Model Testing</td>
<td>45</td>
</tr>
<tr>
<td>5.6.1 Detection Efficiency</td>
<td>45</td>
</tr>
<tr>
<td>5.6.1.1 Layer 1</td>
<td>45</td>
</tr>
<tr>
<td>5.6.1.2 Layer 2</td>
<td>46</td>
</tr>
<tr>
<td>5.6.1.3 Layer 3</td>
<td>46</td>
</tr>
<tr>
<td>5.6.2 Characterization Efficiency</td>
<td>46</td>
</tr>
</tbody>
</table>
### LIST OF TABLES

<table>
<thead>
<tr>
<th>Table</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>4.1</td>
<td>Summary of Monitoring Well Network Characterization Efficiencies, Butler County Landfill</td>
<td>31</td>
</tr>
<tr>
<td>5.1</td>
<td>Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 1)</td>
<td>48</td>
</tr>
<tr>
<td>5.2</td>
<td>Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 2)</td>
<td>48</td>
</tr>
<tr>
<td>5.3</td>
<td>Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 3)</td>
<td>48</td>
</tr>
<tr>
<td>5.4</td>
<td>Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 1)</td>
<td>49</td>
</tr>
<tr>
<td>5.5</td>
<td>Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 2)</td>
<td>49</td>
</tr>
<tr>
<td>5.6</td>
<td>Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 3)</td>
<td>49</td>
</tr>
<tr>
<td>6.1</td>
<td>Summary of Monitoring Well Network Composite Efficiencies</td>
<td>67</td>
</tr>
</tbody>
</table>
LIST OF FIGURES

3.1 Decay of Primary Covering Weight, \( w_{si} \), as a Function of Distance from Contaminant Source, \( D_{si} \); \( D_f \) - Decay Factor

3.2 Hypothetical Advective Envelope; Rectangular Area - Contaminant Source; Thick Lines Extending from Source - Boundaries of Advective Envelope; Thin Lines - Hydraulic Head Contours; Elevations in Units of Length

4.1 Butler County, Ohio Landfill Study Area

4.2 Schematic Geologic Cross-Section through Study Area

4.3 Original Monitoring Well Locations (Circles and Dots), Existing Monitoring Well Locations (Circles), and Observed Steady-State Head Distribution; Elevations in Meters above Mean Sea Level (msl); Arrows Denote Boundaries of Advective Envelope

4.4 Analytical Model Representation, Butler County Landfill; Small Dots - Potential Well Sites; Large Dots - Supply Well Sites; Triangles - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

4.5 Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Butler County Landfill; Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

4.6 Contaminant Nodes Covered by Original Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

4.7 Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

4.8 Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

4.9 MWB Model-Determined Well Sites (Crosses and Large Dots), Butler County Landfill; \( ss = 0.5; D_f = 0.0; w_1 = 0.1; w_2 = 0.9; \) Squares - Landfill Boundary Nodes; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

5.1 Casmalia Study Area (Santa Barbara County, California); 1 Mile = 1.6 Kilometers
5.2 Distribution of On-Site Waste Impoundments, Casmalia Waste Facility; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987a) 51

5.3 Water Supply Well Locations (Large Dots); Labels Indicate Well Identification Number, Well Type (I: Irrigation, D: Domestic, P: Public, S: Stock) and Assigned User Population (in Parentheses) (U.S.G.S. Topographic Map - Casmalia, Guadalupe, Santa Maria, Orcutt (CA) 7.5 Minute Quadrangles, 1982; Elevations in Feet msl; 1 Foot = 0.3048 Meters) 52

5.4 Water Table Contours (Feet msl), Advective Envelopes (Shaded Areas), and Location of Hydrogeologic Cross-Section A-A'; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987b) 53

5.5 Hydrogeologic Cross-Section A-A'; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987b) 54

5.6 Inferred Distribution of Hydraulic Head in Layer 3 at 200 to 300 Feet msl Interval (Contours in Feet msl) and Advective Envelopes (Light Shaded Area) Constructed from Areas of Downward Flux (Dark Shaded Areas) within Facility Boundaries; 1 Foot = 0.3048 Meters 55

5.7 Analytical Model Representation, Casmalia Waste Facility (Layer 1); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 350 Feet (106.7 Meters) 56

5.8 Analytical Model Representation, Casmalia Waste Facility (Layer 2); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 350 Feet (106.7 Meters) 56

5.9 Analytical Model Representation, Casmalia Waste Facility (Layer 3); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 700 Feet (213.4 Meters) 57

5.10 Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Circles - Upgradient Wells; Circles Enclosing Diamonds - Upgradient Wells Receiving Contamination; Circle Enclosing Cross Superimposed on Diamond - Two Original Wells (One Designated for Upgradient Monitoring) at Site Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters) 57

5.11 Contaminant Nodes Covered by Original Well Sites (Crosses), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters) 58
5.20 Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 700 Feet (213.4 Meters)

5.21 Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 700 Feet (213.4 Meters)

5.22 Basic Model-Determined Well Sites (Large Dots), Casmalia Waste Facility (Layer 3); \( w_1 = 0.0; w_2 = 1.0 \) (U.S.G.S. Topographic Map - Casmalia, Guadalupe, Santa Maria, Orcutt (CA) 7.5 Minute Quadrangles, 1982; Elevations in Feet msl; 1 Foot = 0.3048 Meters)
NOTATION

\( C \) = number of wells sited in a hydrostratigraphic interval divided by the number of possible well sites in the interval

\( C_h \) = contaminant hazard factor at water supply well \((1/L^2)\)

\( C_{jk} \) = cost of constructing a well at site \( j \), layer \( k \) (monetary units)

\( D_f \) = MWB model decay factor (dimensionless)

\( D_{pi} \) = distance between node \( i \) and advective envelope (L)

\( D_{si} \) = distance between node \( i \) and contaminant source (L)

\( d_{ijk} \) = shortest distance between nodes \( i \) and \( j \), layer \( k \) (L)

\( E_i \) = value for exposure term, node \( i \) (dimensionless)

\( E_f \) = network detection efficiency (dimensionless)

\( H_i \) = value for hydrogeologic term, node \( i \) (dimensionless)

\( I_k \) = set of nodes in layer \( k \)

\( I_{dk} \) = set of detection nodes, layer \( k \)

\( I_{sk} \) = set of water supply well nodes, layer \( k \)

\( I \) = index of potential well site

\( J_{bk} \) = set of potential well sites along source boundaries, layer \( k \)

\( J_k \) = set of potential well sites, layer \( k \)

\( J_{pk} \) = set of nodes occupied by pre-existing wells, layer \( k \)

\( J_{zk} \) = set of potential well sites in zone \( z \), layer \( k \)

\( J_{1k} \) = set of potential well sites downgradient from contaminant source, layer \( k \)

\( J_{2k} \) = potential well sites upgradient from contaminant source, layer \( k \)

\( j \) = areal index of potential well site

\( K \) = set of layers

\( k, m \) = hydrostratigraphic interval (HSI) indices

\( M_{ik} \) = set of potential well sites, layer \( k \), located at a distance between \( S_k \) and \( T_k \) from node \( i \)

\( N_k \) = number of possible well sites, layer \( k \)

\( N_{1k} \) = set of potential well sites, layer \( k \), located within a distance \( S_k \) of node \( i \)

\( N_{wb} \) = number of wells (excluding upgradient wells) in a network with concentrations above background

\( N_{wt} \) = total number of wells (excluding upgradient wells) in a network

\( P \) = total number of wells to be located

\( P_k \) = total number of wells to be located in layer \( k \)

\( P_k^{(\text{min})} \) = minimum number of wells to be located in layer \( k \)

\( P_s \) = population served by water supply well

\( P_{zk}^{(\text{max})} \) = maximum number of wells to be located in zone \( z \), layer \( k \)
$P_{zk}(\text{min})$ = minimum number of wells to be located in zone $z$, layer $k$

$P_{1k}$ = total number of wells to be located in downgradient zone, layer $k$

$P_{2k}$ = total number of wells to be located in upgradient zone, layer $k$

$R$ = total available monetary resources (monetary units)

$S_k$ = primary distance threshold, layer $k$ (L)

$S_c$ = total value that a well sited on a source-distant node would contribute to the model objective function if coverage was governed by the standard covering weights (dimensionless)

sp = standard primary covering weight (dimensionless)

ss = standard secondary covering weight (dimensionless)

$T_k$ = secondary distance threshold, layer $k$ (L)

$v_{ik}$ = one if node $i$, layer $k$ is part of a well nest; $v_{jk} = zero$ otherwise (used in MWB model)

$v_{ik}$ = one if node $j$, layer $k$ is part of a well nest; $v_{jk} = zero$ otherwise (used in basic model)

$v_{i*}$ = scaled ground water velocity parameter, node $i$ (dimensionless)

$W_{ik}$ = nodal weight (node $i$, layer $k$) in MWB model (dimensionless)

$W_{jk}$ = nodal weight (node $i$, layer $k$) in basic model (dimensionless)

$W_p$ = sum of $W_i$ values for perimeter nodes (perimeter nodes designated by index $p$) (dimensionless)

$w_{sik}$ = primary covering weight for node $i$ (relative weight attached to coverage between distances of zero to $S_k$), layer $k$ (dimensionless)

$w_{lik}$ = secondary covering weight for node $i$ (relative weight attached to coverage between distances of $S_k$ to $T_k$), layer $k$ (dimensionless)

$w_1$ = detection term partitioning weight (dimensionless)

$w_2$ = exposure term partitioning weight (dimensionless)

$x_{jk}$ = one if a well is sited at node $j$, layer $k$; $x_{jk} = zero$ otherwise

$y_{ik}$ = one if node $i$, layer $k$ is covered by the closest well in layer $k$ within a distance of $S_k$; $y_{ik} = zero$ otherwise

$Z$ = value of objective function (dimensionless)

$Z_k$ = set of zones in layer $k$

$Z_1$ = value of objective function in basic model problem formulation 1 (dimensionless)

$Z_2$ = value of objective function in basic model problem formulation 2 (dimensionless)

$z$ = zone index

$z_{ik}$ = one if node $i$, layer $k$ is covered by the closest well in layer $k$ at a distance between $S_k$ and $T_k$; $z_{ik} = zero$ otherwise

$\omega$ = weight varied between zero and one (dimensionless)
1. INTRODUCTION

Ground water quality monitoring is an important task in aquifer protection and ground water management. Accurate and timely information on the spatial distribution of physical, chemical, and biological properties of ground water is essential in the formulation of corrective action plans and basin-wide environmental management strategies for aquifers. The successful attainment of this information is highly dependent on the monitoring well configuration from which samples are collected. Ground water quality monitoring network design involves selection of the locations of sampling sites and sampling frequency. Network design methodologies developed in previous studies are primarily applicable to the site-specific or local scale (i.e., length of model domain on the order of $10^1 - 10^2$ m), and generally have been applied to hypothetical or relatively simple aquifers. These methodologies are not applicable to complex, three-dimensional aquifers, nor to regional scale contamination problems where the extent of potential pollution occurs over a much larger area, possibly including areas of ground water withdrawal for consumptive use. This study addresses the problem of ground water monitoring network design for contaminant facilities that have the potential for generating pollution problems of a regional scale. Ground water monitoring network design models were formulated, solved, and tested for two different case studies, each with contaminant sources of sufficient magnitude to generate regional scale ground water contamination. Pre-existing and model-derived monitoring configurations were tested by comparing well sites to contaminant distributions predicted by numerical mass transport models.
2. LITERATURE REVIEW

The four broad approaches to ground water monitoring network design are (Loaiciga et al. 1992) (1) hydrogeologic, (2) simulation, (3) variance-based, and (4) optimization. A methodology for network design may include components of more than one approach. Nevertheless, the classification provides a useful means for grouping a number of relevant research ideas.

2.1 Hydrogeologic Approach

The term "hydrogeologic approach" is used herein to describe the case where the network is designed on the basis of calculations and judgments made by the hydrogeologist without the use of quantitative, mathematical methods. Simulation, variance-based, and optimization approaches require the use of mathematical or statistical models and constitute distinctly quantitative techniques. In the hydrogeologic approach, the locations of sampling sites are determined by the hydrogeologic conditions near the source of contamination. The Resource Conservation and Recovery Act (RCRA) guidelines for ground water monitoring (U.S. Environmental Protection Agency 1986) require, at a minimum, four ground water monitoring wells: one well upgradient and three downgradient from the source of contamination.

The hydrogeologic approach has been applied primarily to local scale settings. The main objective of the approach is the early detection of contaminants in the subsurface (Everett 1980). The ultimate configuration of monitoring wells is subject to the investigator's understanding of (1) the key properties of the ground water flow system, (2) how these properties dictate the movement of contaminant, and (3) what constitutes an "optimal" monitoring well configuration given probable contaminant migration pathways. Due to ease of implementation, this approach is widely used in practice. However, in a purely qualitative approach to monitoring network design, there is no provision for quantifying the relative value of numerous potential sampling sites.

2.2 Simulation Approach

Depending on data availability and aquifer complexity, simulation approaches are conducted in a deterministic or statistical manner. Deterministic approaches predict plume geometry from a specified set of parameter input values. Effective results are contingent upon detailed field data which accurately define spatial variations in aquifer parameters. In statistical simulation approaches, parameters of a porous medium are modeled as random fields (e.g., hydraulic conductivity as a log-normal distribution). By generating multiple synthetic distributions of parameters such as hydraulic conductivity, for each of which there is a corresponding contaminant distribution, it is possible to determine the statistical properties of mass transport in an aquifer and the detection capability of a monitoring network. Examples of statistical simulation approaches to ground water quality monitoring network design include Gilbert (1987), Massman and Freeze (1987a; 1987b), Meyer and Brill (1988), Ahlfeld and Pinder (1988), and Meyer et al. (1989).
The statistical simulation approach is computationally intensive. Typically, several hundred contaminant distributions are generated to determine the locations of only a few well sites. Each contaminant distribution is obtained from a realization of one or more aquifer parameters which are drawn from an assumed distribution. Field or laboratory measurements are required to determine appropriate parameter distributions, or to verify the adequacy of an assumed distribution. Integer programming models must then be formulated and solved to derive optimal well configurations given a network of hundreds of potential sites and numerical model-generated contaminant distributions. Numerical modeling of contaminant transport, especially in three dimensions, is considerably more difficult than simulation of ground water flow. Transport modeling not only is more vulnerable to numerical errors such as numerical dispersion and artificial oscillation, but also requires much more computer memory and execution time, making it impractical for many field applications. Most of the advanced mass transport models require mainframe computing systems. Even given the necessary computational hardware, the time it takes to run an extended simulation can be excessive, especially for complex aquifers and multiple "stress" periods. The generation of multiple contaminant distributions, each corresponding to some realization of a set of statistical aquifer parameters, is impractical for many problems.

2.3 Variance-Based Approaches

Variance-based approaches to ground water quality monitoring network design can be categorized as global or variance-reduction. Global approaches employ indices for the performance of a monitoring network which operate over the entire sampling domain. Such indices include the average or maximum variance of estimation. The goal of the global method is to identify the best pattern (e.g., square, triangular, or other geometric arrangements) and the best density of sampling sites (Olea 1984; Yfantis et al. 1987; Christakos and Olea 1988). Global performance standards are generally inconsistent 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 sub region of the study area.

The variance-reduction approach involves a methodical search for the number and locations of sampling sites that minimize the variance of estimation error of the variable of interest, such as the concentration of a pollutant (Bastin et al. 1984; Rouhani 1985; ASCE Task Committee on Geostatistical Techniques in Geohydrology 1990a; 1990b). The development of a ground water monitoring network configuration starts with a number of existing well sites to which sites are added, one at a time. Wells are added such that each new well site produces the largest possible reduction (i.e., out of all possible sites) in the variance of contaminant concentration estimation error. Sampling sites are added until the variance of estimation error can no longer be (or is only marginally) reduced, or when the marginal gain in statistical accuracy is outweighed by other constraints, such as limited resources.

Variance-reduction approaches have also been used to optimize the parameter estimation process for numerical transport modeling (Knopman et al. 1991). Estimates of parameters such as dispersion coefficients are
obtained by calibrating a model to a set of observed concentrations. The locations and timing of concentration observations influence parameter estimation for a system. The goal of the network design problem is to minimize the variance of parameter estimates. Low variances in parameter estimates are obtained by locating wells at points of high sensitivity, where the change in concentration resulting from a change in a given parameter is high (Knopman and Voss 1987).

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 studies (ASCE Task Committee on Geostatistical Techniques in Geohydrology 1990a; 1990b) indicate that the variance-reduction analysis is best suited for the determination of additional sampling sites.

2.4 Optimization Approach

In the optimization approach, the ground water quality monitoring network design problem is formulated within a mathematical programming framework. There is a well-defined objective function, such as minimizing the probability of human exposure to toxic substances in ground water, or maximizing the statistical accuracy associated with estimates of measured ground water quality properties (Hsueh and Rajagopal 1988; Knopman and Voss 1988; Loaiciga 1989). Constraints are explicitly considered while optimizing the objective function. Such constraints might include resource constraints, governing equations of physical processes, and statistical constraints (Hsueh and Rajagopal 1988; Loaiciga 1989; Loaiciga and Hudak 1989; Scheibe and Lettenmaier 1989).

Because a key output of the optimization approach is the location of sampling sites, the corresponding mathematical programming problem usually involves the use of binary integer variables that reflect either the placement or the absence of an actual sampling site at each potential sampling location (Hsueh and Rajagopal 1988; Nemhauser and Wolsey 1988; Loaiciga 1988; 1989). The integer programming formulation has been widely used in location theory applications (Toregas et al. 1971; Toregas and ReVelle 1972; 1973; Church and ReVelle 1974; Meyer and Brill 1988; Hudak and Loaiciga 1992).

2.5 Summary

The approaches outlined above are generally applicable to network design at the local scale (hydrogeologic and simulation approaches) or to augmenting existing networks for the purpose of contaminant distribution characterization (variance-based approaches). Optimization approaches have been developed for regional scale settings (Hsueh and Rajagopal 1988) where sampling sites are selected from a set of existing water supply wells. However, these approaches are not relevant to the problem of locating new sampling sites in the regional vicinity of a well-defined pollution source. The approach developed in this study is applicable to the problem of deriving a network of new sampling sites in an uncontaminated aquifer at risk of regional scale...
contamination from an overlying waste facility. A regional scale problem is used in this context to designate a case where the area of potential contamination extends a distance of at least $10^3$ m from the contaminant source.
3. MODEL THEORY AND DEVELOPMENT

Contaminant detection is a fundamental objective of ground water monitoring network design. On a regional scale, it is essential that monitoring be conducted at (1) areas likely to become contaminated near and downgradient of the contaminant source, and (2) water supply wells at risk of contamination. Monitoring at water supply wells is required due to the severe consequences that may result from contamination at such locations and the resultant need for direct confirmation of a potential problem. The value of a candidate monitoring site can be assessed with regard to either of (1) or (2) above (respectively, referred to as "detection value" and "exposure hazard value"). A comprehensive regional scale network design methodology should effectively quantify spatial variations in monitoring value and utilize this information to determine the optimal locations of sampling sites. The approach taken here is to quantify key parameters that affect detection and exposure hazard value throughout the area potentially affected by a source of contamination.

First, we partition a "primary area" of the model domain into a field of candidate monitoring sites. The primary area includes the contaminant source and areas potentially affected by the migrating contaminant, and extends to hydrogeologic boundaries (such as regional discharge areas or no-flow boundaries). Nodes within the primary area, referred to as "detection nodes," are arranged in a regular geometric pattern. Node spacing is based on the minimum width of potential contaminant migration outlets at facility boundaries and computational considerations. Existing water supply wells are also treated as potential monitoring sites. These wells are generally located outside the primary area, within a "secondary area."

Each candidate well site is assigned a weight quantifying monitoring value. From a field of candidate nodes and associated weights, a mathematical programming model selects an optimal arrangement of monitoring well sites. We seek two key properties through the selection process: preferential location of well sites at or near points likely to become contaminated, and sufficient inter-well separation to allow an adequate characterization of the extent of migrating contaminant. In order to ensure monitoring throughout a multi-layered flow system, it is necessary to impose constraints requiring a minimum number of wells in each layer (layer constraints), upgradient monitoring (a form of the "zonal constraint" introduced by Church (1990)) and a specified total number of wells.

Two mathematical programming models were developed: a basic integer programming model (basic model), and a modified version of the Weighted Benefit Maximal Coverage (WBMC) Model (Church and Roberts 1983). The modified version of the WBMC model developed in this study is termed the Modified Weighted Benefit or MWB model. The basic model selects as sampling sites the nodes with the highest weight values. The MWB model is a more sophisticated model that considers the pattern of well selection in optimizing a model objective function.
3.1 Basic Model

The formulation for the basic model is

\[
\text{Max } Z = \omega \sum_{k \in K} \sum_{j \in J_k} W_{jk} x_{jk} + (1 - \omega) \sum_{k \in K} \sum_{j \in J_k} W_{jk} v_{jk} \tag{3.1}
\]

subject to

\[
\sum_{j \in J_k} x_{jk} \geq P_k \text{ (min) for each } k \in K \tag{3.2}
\]

\[
\sum_{j \in J_k} x_{jk} \leq P_k \text{ (max) for each } z \in Z_k, k \in K \tag{3.3}
\]

\[
\sum_{j \in J_k} x_{jk} \geq P_k \text{ (min) for each } z \in Z_k, k \in K \tag{3.4}
\]

\[
\sum_{k \in K} \sum_{j \in J_k} x_{jk} = P \tag{3.5}
\]

\[
v_{jk} \leq x_{jk} \text{ for each } j \in J_k, k \in K \tag{3.6}
\]

\[
v_{jk} \leq (\sum_{m \in K} x_{jm}) - 1 \text{ for each } j \in J_k, k \in K \tag{3.7}
\]

\[
x_{jk} = (0,1) \text{ for each } j \in J_k, k \in K \tag{3.8}
\]

\[
v_{jk} = (0,1) \text{ for each } j \in J_k, k \in K \tag{3.9}
\]

where

\(j\) = areal (horizontal) index of potential well site

\(k, m\) = hydrostratigraphic interval (HSI) indices (The index \(m\) is used for the vertical interaction constraint, see below.)

\(z\) = zone index

\(J_k\) = set of potential well sites, layer \(k\)

\(J_{zk}\) = set of potential well sites in zone \(z\), layer \(k\)

\(K\) = set of layers

\(Z_k\) = set of zones in layer \(k\)

\(W_{jk}\) = weight for node \(j\), layer \(k\) (explained in Section 3.3)

\(x_{jk}\) = one if a well is installed at site \(j\), layer \(k\); \(x_{jk}\) = zero otherwise (The notation \(x_{jm}\) is used for the vertical interaction constraint.)
The first term in the objective function (3.1) is the sum of nodal weights for all nodes in each hydrostratigraphic interval (HSI) of a multi-layered system. If a well is sited at a node, it contributes the corresponding weight value to the objective function. The second term in (3.1), referred to as the "vertical interaction term," is the sum of weights for nodes that are part of well nests (i.e., wells located side by side, but screened within different hydrostratigraphic intervals). The values for the two terms in the objective function depend on the partitioning in emphasis established by the weight \( \omega \). Constraint (3.2) ensures that a minimum number of wells are located in each HSI, and constraints (3.3) and (3.4) are zonal constraints. The zonal constraints ensure that the number of wells allocated to each zone in a layer is between specified minimum and maximum values. Such zones could correspond to areas specified for upgradient and downgradient monitoring. Constraint (3.5) establishes the total number of wells to be located throughout a model domain. Constraints (3.6) and (3.7), referred to as vertical interaction constraints, define conditions necessary for well nesting; (3.6) specifies that a node cannot be part of a well nest unless a well is sited at that node, and (3.7) defines a well nest as a cluster of two or more wells at a single areal location \( j \). Both \( x_{jk} \) and \( v_{jk} \) are binary integers. However, for a given \( j \) and \( k \), \( v_{jk} \) can only equal one if (1) a well is screened at site \( (j,k) \) (i.e., \( x_{jk} \) equals one), and (2) one or more wells are screened above or below HSI \( k \), at the areal location \( j \). Finally, constraints (3.8) and (3.9) impose binary integer conditions on model variables.

If it is not desirable to explicitly incorporate a vertical interaction term, and if a specified number of wells is assigned to each HSI, the basic model formulation can be decomposed to a series of separate layer applications. The form of the layer-decomposed model, with a specified number of wells in each zone, is

For each HSI \( k \):

\[
\text{Max } Z = \sum_{j \in J_k} W_{jk} x_{jk}
\]  

(3.10)

subject to

\[
\sum_{j \in J_{zk}} x_{jk} = P_{zk} \quad \text{for each } z \in Z_k
\]  

(3.11)

\[
\sum_{j \in J_k} x_{jk} = P_k
\]  

(3.12)
\[ x_{jk} = (0,1) \text{ for each } j \in J_k \quad (3.13) \]

The quantity \( P_k \) is the specified number of wells to be located in a given layer \( k \) (other variables retain earlier definitions). The layer-decomposed model is easier to implement and solve because it includes fewer decision variables and constraints.

Each of the general and layer-decomposed models requires the specification of a total number of wells to be located. This number could be constrained by regulatory requirements or budget constraints. Budgetary restrictions can be directly incorporated into the model by adding a constraint of the form (using general model format):

\[
\sum_{k \in K} \sum_{j \in J_k} C_{jk} x_{jk} \leq R \quad (3.14)
\]

where
- \( C_{jk} = \text{cost of constructing a well at site } j, \text{ layer } k \)
- \( R = \text{total available monetary resources} \)

However, with constraint (3.14), a low value of \( R \) may cause a sparse, ineffective detection network. Furthermore, the form of the constraint decreases the likelihood of model solution via relaxed linear programming and thus increases computational requirements.

If two distinct zones are identified, corresponding to regions of upgradient and downgradient monitoring, the layer-decomposed model given by (3.10) through (3.13) can be further reduced to two smaller, zonal problems:

For each HSI \( k \):

\[ \text{Problem 1 - Downgradient} \]

\[ \text{Max } Z_1 = \sum_{j \in J_{ik}} W_{jk} x_{jk} \quad (3.15) \]

subject to

\[ \sum_{j \in J_{ik}} x_{jk} = P_{1k} \quad (3.16) \]

\[ x_{jk} = (0,1) \text{ for each } j \in J_{ik} \quad (3.17) \]
Problem 2 - Upgradient

\[ \text{Max } Z_2 = \sum_{j \in J_{2k}} W_{jk} x_{jk} \]  
\[ \text{subject to} \]  
\[ \sum_{j \in J_{2k}} x_{jk} = P_{2k} \]  
\[ x_{jk} = (0,1) \text{ for each } j \in J_{2k} \]  
\[ (3.18) \]
\[ (3.19) \]
\[ (3.20) \]

In Problem 1, the set of potential well sites consists of all candidate nodes, excluding upgradient sites, in layer \( k \). From this set, represented by the symbol \( J_{1k} \), \( P_{1k} \) nodes are selected as monitoring sites. In Problem 2, \( P_{2k} \) wells are allocated to the set of nodes designated as potential upgradient monitoring sites (i.e., the set \( J_{2k} \)).

Integer programming techniques such as branch-and-bound (Land and Doig 1960) can be used to solve the general formulation given by (3.1) through (3.9). Branch-and-bound is a structured enumeration procedure in which a fraction of the total set of feasible solutions to a problem are examined. Solution algorithms can be found in optimization software packages such as LINDO (Schrage 1987; 1991) and GAMS (Brooke et al. 1988). Problems 1 and 2 represent single constraint zero-one knapsack problems (Dantzig 1963) with cost coefficients equal to one. These models can be solved by solution methods that have been developed for the single constraint knapsack problem structure (Dantzig 1957; 1963) and branch-and-bound algorithms by authors such as Kolesar (1967). In general, single constraint knapsack algorithms selectively choose the variables with the highest weights while satisfying a constraint on total cost (a cost is incurred for each variable added to the solution). For the case of a uniform cost per variable equal to one, the solution is obtained by choosing the specified number of variables with the highest weights. Thus, Problems 1 and 2 can be solved with sort functions available in statistical software packages. Given a set of nodes with assigned weight values, these functions arrange the set in order of ascending or descending values, and the specified number of nodes with the highest values can be extracted from the proper end of the sequence.

Important features of the basic model include relative ease of implementation and solution, assignment of monitoring wells to each HSI, preferential location of wells in high susceptibility areas near the source of contamination, upgradient monitoring, and inclusion of existing supply wells among the set of potential monitoring sites. A potential problem with this model is the possibility of inadequate well coverage throughout a model domain due to well clustering near the source of contamination, where the \( W_{jk} \) coefficients would generally be highest. For a potentially widespread contamination problem, it is inappropriate to site wells only at the boundaries of the contaminant source. There are several reasons why such a configuration could be inappropriate.

1. There is a distinct possibility that contamination could migrate undetected beyond a band of wells located along the source boundaries. This possibility is enhanced by the relative narrowness of the plume in the initial stages of
evolution. As a result of mechanical dispersion and molecular diffusion, contaminant plumes tend to increase in size downgradient from the source of contamination. Monitoring wells sited away from a source can be effective for detecting contamination, especially if the plume has attained significant size with downgradient transport. (2) If a plume migrates beyond monitoring wells sited solely at the contaminant source, there is no provision for tracking the movement and growth of the plume without additional downgradient monitoring wells. (3) A network consisting of wells sited only at source boundaries cannot adequately characterize a plume that has evolved from a facility, even in the event that it is detected by such wells. (4) Upgradient wells located away from the contaminant source boundaries must be sited to allow comparison of contaminant concentrations in downgradient wells with ambient, uncontaminated ground water conditions. (5) Alluvial channels or other preferential pathways for contaminant migration extending beyond the boundaries of the contaminant source may be more susceptible to inundation by a migrating plume than relatively low hydraulic conductivity aquifer materials immediately adjacent to the source.

The problem of well clustering can be avoided by models that generate sufficient inter-well separation distances. Covering-based location models are well suited to generating configurations which satisfy this objective. The Maximal Covering Location Model (MCLM) (Church and ReVelle 1974) sites a fixed number of facilities on a network of nodes to maximize coverage (demand covered) within a distance threshold. The threshold defines the maximum distance within which a demand node can be served by a facility. In general, the most efficient distribution of facilities is one characterized by some degree of spreading or dispersion. The distance threshold effectively dictates the degree of separation between adjacent facilities; with larger thresholds, facilities are located further apart to avoid coverage overlap. An analogy can be made between a field of nodes within which monitoring wells are to be placed and a network of demand nodes in a facility location problem. Unlike a service facility, a monitoring well only monitors contaminant within a local screened interval. Although it would be inappropriate to suggest that a well characterizes all contaminant within a distance threshold greater than zero, the distance threshold can be employed such that individual wells sample areas of high contamination susceptibility (i.e., areas likely to become contaminated) while the network as a whole maintains some degree of areal coverage.

3.2 Modified Weighted Benefit Coverage Model (MWB Model)

The MWB Model is a modification of the Weighted Benefit Maximal Coverage (WBMC) Model (Church and Roberts 1983) which, in turn, is a modification of the Maximal Covering Location Model (MCLM) (Church and ReVelle 1974). As does the basic model, the MWB model maximizes the sum of nodal weights contributed to the model objective function. However, in the MWB model, sited wells (excluding water supply wells) cover (contribute the values of) weights associated with both the corresponding node and adjacent neighboring nodes.
The MWB model formulation is

\[
\text{Max } Z = \alpha \left[ \sum_{k \in K, \ell \in \Lambda_k} (w_{ik} W_{ik} \bar{y}_{ik} + w_{ik} W_{ik} \bar{z}_{ik}) + \sum_{k \in K, \ell \in \Lambda_k} W_{ik} \bar{y}_{ik} \right] + (1 - \alpha) \sum_{k \in K, \ell \in \Lambda_k} W_{ik} \bar{v}_{ik} 
\]

subject to

\[
\sum_{j \in N_{ik}} x_{jk} \geq y_{ik} \quad \text{for each } i \in I_k, k \in K
\]

\[
\sum_{j \in M_{ik}} x_{jk} \geq z_{ik} \quad \text{for each } i \in I_k, k \in K
\]

\[y_{ik} + z_{ik} \leq 1 \quad \text{for each } i \in I_k
\]

\[x_{jk} = 0 \quad \text{for each } j \in J_k, k \in K
\]

\[
\sum_{j \in J_k} x_{jk} \geq P_{ik} \text{ (min)} \quad \text{for each } k \in K
\]

\[
\sum_{j \in J_k} x_{jk} \leq P_{ik} \text{ (max)} \quad \text{for each } z \in Z_k, k \in K
\]

\[
\sum_{j \in J_k} x_{jk} \geq P_{ik} \text{ (min)} \quad \text{for each } z \in Z_k, k \in K
\]

\[
\sum_{k \in K, j \in J_k} x_{jk} = P_k
\]

\[v_{ik} \leq y_{ik} \quad \text{for each } i \in I_k, k \in K
\]

\[v_{ik} \leq (\sum_{m \in K} y_{im}) - 1 \quad \text{for each } i \in I_k, k \in K
\]

\[y_{ik} = (0,1) \quad \text{for each } i \in I_k, k \in K
\]

\[z_{ik} = (0,1) \quad \text{for each } i \in I_k, k \in K
\]

\[x_{jk} = (0,1) \quad \text{for each } j \in J_k, k \in K
\]

\[v_{ik} = (0,1) \quad \text{for each } i \in I_k, k \in K
\]

where

\[i = \text{index of potential well site}
\]

\[j = \text{areal (horizontal) index of potential well site}
\]

\[k,m = \text{layer (HSI) indices (The index } m \text{ is used for the vertical interaction constraint, see below.)}
\]

\[z = \text{zone index}
\]

\[I_k = \text{set of nodes in layer } k
\]
\( I_{dk} = \) set of detection nodes, layer \( k \)
\( I_{sk} = \) set of water supply well nodes, layer \( k \)
\( J_k = \) set of potential well sites, layer \( k \)
\( J_{bk} = \) set of potential well sites along source boundaries, layer \( k \)
\( J_{zk} = \) set of potential well sites in zone \( z \), layer \( k \)
\( K = \) set of layers
\( Z_k = \) set of zones in layer \( k \)
\( W_{ik} = \) weight for node \( i \), layer \( k \) (explained in Section 3.3)
\( d_{ijk} = \) shortest distance between nodes \( i \) and \( j \), layer \( k \)
\( N_{ik} = \) \( \cup \{d_{ijk} \leq S_k\} \)
\( M_{ik} = \) \( \cup \{S_k < d_{ijk} \leq T_k\} \)
\( S_k, T_k = \) distance thresholds in layer \( k \) used in coverage calculation where \( S_k \leq T_k \) (For this application, \( S_k = 0 \), indicating direct coverage, and \( T_k = \) one grid interval.)
\( y_{ik} = \) one if node \( i \), layer \( k \) is covered by the closest well with a distance \( S_k \); \( y_{ik} = 0 \) otherwise (The notation \( y_{im} \) is used with the vertical interaction constraint.)
\( z_{ik} = \) one if node \( i \), layer \( k \) is covered by the closest well at a distance between \( S_k \) and \( T_k \) (i.e., covered within one grid interval); \( z_{ik} = 0 \) otherwise
\( w_{sik} = \) relative weight attached to coverage between distances of zero to \( S_k \) (i.e., direct coverage)
\( w_{tik} = \) relative weight attached to coverage between distances of \( S_k \) to \( T_k \) (coverage within one grid interval)
\( x_{ijk} = \) one if a well is installed at site \( j \), layer \( k \); \( x_{ijk} = 0 \) otherwise (The notation \( x_{jm} \) is used with the vertical interaction constraint.)
\( v_{ik} = \) one if a well at site \( i \), layer \( k \) is part of a well nest; \( v_{jk} = 0 \) otherwise
\( P_{k}(\text{min}) = \) minimum number of wells to be located in layer \( k \)
\( P_{zk}(\text{max}) = \) maximum number of wells to be located in zone \( z \), layer \( k \)
\( P_{zk}(\text{min}) = \) minimum number of wells to be located in zone \( z \), layer \( k \)
\( P = \) total number of wells to be located
\( \omega = \) weight varied between zero and one

The objective function, (3.21), consists of two primary terms: a covering term and a vertical interaction term. The first term (covering term) consists of three smaller terms (from left to right, terms 1A, 1B, 1C). For a sited well (excluding supply well sites), term 1A represents some fraction of the weight for the directly covered node (i.e., the node at which the well is sited). Term 1B represents a fraction of the weights associated with the neighboring nodes. The parameters \( w_{sik} \) and \( w_{tik} \) define the fractions of the nodal weight values (\( W_{ik} \)) contributed to the objective function. If a water supply well is selected as a monitoring site, the full value of the weight for that site is contributed to the objective function (term 1C). Note that terms 1A and 1B contain two different types of weights.
The upper-case $W_{ik}$ weights quantify monitoring value at each candidate node. The lower-case $w_{si k}$ and $w_{ti k}$ weights are the covering weights: $w_{si k}$ is the primary covering weight; $w_{ti k}$ is the secondary covering weight. The $i$ index indicates that the values of the covering weights for a given layer $k$ depend on the locations of potential well sites with respect to the contaminant source.

Constraints (3.22) and (3.23) define coverage within ranges of distance; (3.22) pertains to all candidate sites; (3.23) pertains only to detection nodes (within the primary area). Constraint (3.24) ensures that coverage of a node is calculated in only one way (i.e., the weight value for a node can be contributed to the objective function by virtue of the presence of only one well). Constraint (3.25) ensures that landfill boundary nodes are not chosen as monitoring sites. The minimum numbers of wells to be allocated to each layer are specified by constraint (3.26), and constraints (3.27) and (3.28) are zonal constraints. The total number of wells allocated to the entire model domain is given by constraint (3.29). Constraints (3.30) and (3.31) are vertical interaction constraints, and the remaining constraints require that model variables be binary integer. Note that constraints (3.32) and (3.33) are defined separately for the $y_i$ and $z_i$ variables. The variable $y_i$ represents direct coverage and operates over all candidate nodes; $z_i$ represents secondary coverage and pertains only to detection nodes.

The concept of covering operates on a layer-by-layer basis; there is no covering across layers. In the discussion to follow, the subscript $k$ has been omitted. A well placed on a detection node covers the node, and a fraction (represented by the covering weight $w_{si}$) of the corresponding $W_i$ value is contributed to the objective function. Neighboring nodes at one grid spacing are also covered, and a smaller fraction (represented by the covering weight $w_{ti}$) of the corresponding $W_i$ values are contributed to the objective function. The covering weights are chosen such that $w_{si} > w_{ti}$. The case where $w_{si} = w_{ti}$ represents a situation where a well covers the corresponding node and neighboring nodes in equivalent fractions, thereby contributing the same portion of the $W_i$ values to the objective function. In this case, for the purpose of optimizing the model objective function, it is generally disadvantageous to site wells at neighboring nodes. Therefore, a condition of $w_{si} = w_{ti}$ leads to well separation. If $w_{si}$ is significantly larger than $w_{ti}$, it is more advantageous to cover nodes directly (versus by a neighboring well), particularly those with high $W_i$ values. This situation leads to well clustering.

A monitoring configuration with wells spaced close together near the contaminant source and further apart downgradient is potentially effective for both detection and characterization of migrating contamination. This pattern can be achieved by allowing the covering weights, $w_{si}$ and $w_{ti}$, to vary in space. In the original Weighted Benefit Maximal Coverage Model of Church and Roberts (1983), the covering weights are constant. A procedure for varying the covering weights follows.

1. Specify $w_{si}$ and $w_{ti}$ for a reference node near the contaminant source, where $w_{si} \geq w_{ti}$. Designate these as the "standard primary" and "standard secondary" covering weights (sp and ss, respectively). By convention, the standard primary covering weight is set equal to 1.0. For example, choose 1.0 and 0.5.
(2) For all other nodes, calculate $w_{si}$ as a function of $D_{si}$, the distance from node $i$ to the contaminant source. The rate of decay of $w_{si}$ with distance is user-specified, and may be related to the nature of contaminant spreading expected to be characteristic to the ground water flow system. This rate is quantified by the model parameter $D_f$, the "decay factor." The expression relating $w_{si}$, $D_f$ and distance between a node $i$ and the contaminant source, $D_{si}$, is

$$w_{si} = \left[ \frac{D_f}{D_{si} - D_{si}^{(min)}} \right] + 1$$

(3.36)

where

- $D_f$ = decay factor (0 $\leq D_f \leq 1$)
- $D_{si}^{(max)}$ = maximum $D_{si}$ value among all nodes
- $D_{si}^{(min)}$ = minimum $D_{si}$ value among all nodes

In a homogeneous system dominated by high average linear velocities and mechanical dispersion, diffusive spreading of contaminant is relatively unimportant, plumes are narrower, and a small well spacing would be warranted to optimize the possibility of contaminant release detection. Therefore, it would be appropriate to implement a model where $w_{si}$ decays at a low rate from its maximum value near the source. A low rate of decay would result in a condition of $w_{si} > w_{ti}$ both near and away from the contaminant source, which would lead to well clustering. The distance over which a condition of $w_{si} > w_{ti}$ is maintained is a function of the decay factor, $D_f$. The smaller the decay factor, the greater the distance away from the contaminant source over which the condition $w_{si} > w_{ti}$ is maintained. In cases where narrow contaminant plumes are expected, well clustering is warranted, and $D_f$ should be set to a low value, near zero. If relatively broad contaminant plumes are expected (e.g., in aquifers with a low hydraulic conductivity where contaminant spreading is dominated by molecular diffusion), a larger well spacing should be utilized by setting $D_f$ to a high value, near one. The derivation of Equation (3.36) is discussed in greater detail in the next section.

(3) Given a $w_{si}$ value for a node, calculate the corresponding $w_{ti}$ value as follows:

$$w_{ti} = S_c - W_i w_{si}$$

(3.37)

where

- $S_c$ = total value that a well sited on the source-distant node would contribute to the objective function if coverage was governed by the standard covering weights
- $W_p$ = sum of $W_i$ values for the neighboring (perimeter) nodes
In this manner, \( w_{ti} \) is calculated such that the total coverage contributed by a well is the same as if the standard covering weights were used. This property is necessary so that the choice of \( w_{si} \) and \( w_{ti} \) influence the pattern of a monitoring configuration, but not which nodes are chosen on the basis of distance from the contaminant source. The latter is determined by the nodal weights, \( W_i \).

(4) If steps (1) through (3) result in \( w_{ti} > w_{si} \) for a source-distant node, re-calculate the covering weights by simultaneously solving (3.37) and

\[
W_{st} = W_{tt}
\]

(3.38)

The effect of the decay factor, \( D_f \) on \( w_{si} \) is graphically illustrated in figure 3.1. The primary covering weight, \( w_{si} \) varies continuously from a maximum value of one at the reference node (a node, arbitrarily chosen, at the minimum possible distance, \( D_{si}(\text{min}) \) from the contaminant source) to a minimum value between zero and one, inclusive, at the node furthest from the source (i.e., at a distance \( D_{si}(\text{max}) \)). The rate of decay of \( w_{si} \) between \( D_{si}(\text{min}) \) and \( D_{si}(\text{max}) \) is governed by the decay factor, \( D_f \) according to Equation (3.36). Note that in the application of the MWB model, \( w_{si} \) is never less than \( W_u \). As \( w_{ti} \) is calculated for each node successively outward from the origin, or source of contamination (see figure 3.2), a condition of \( w_{ti} > w_{si} \) leads to a re-calculation of both covering weights according to Equations (3.37) and (3.38).

The significance of (3.21) is that it leads to the siting of wells at high priority locations while maintaining inter-well separation. This separation results from interactions embedded in the model whereby wells can influence the fractions of weights contributed to the objective function from more than one site. Model capabilities for siting wells at points of high priority and inducing inter-well separation lead to monitoring networks suitable for both contaminant detection and plume characterization on a regional scale. The MWB model objective function (3.21) contrasts to performance criteria utilized in alternative approaches to network design which are better suited to characterization at the local scale. For example, variance-reduction approaches are geared toward augmenting existing networks and are not suited to the problem of siting wells in uncontaminated aquifers at risk of contamination from an overlying waste facility. There are limitations to variance-reduction approaches which render them unsuited to the problem addressed in this study. (1) The structural analysis which quantifies spatial correlation (Journel and Huijbregts 1978) is derived from information obtained from existing monitoring sites. In many cases, few or no such sites may be available. (2) Variance-reduction approaches generally assign 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 monitor at locations where contaminant levels are high. (3) Many variance-reduction approaches seek to minimize estimation error within a local estimation area (Loaiciga 1989). This will lead to inadequate characterization over a broader, regional scale. (4) As stated in Section 2, the statistical nature of the approach precludes its applicability to complex aquifer systems.
The MWB model can also be layer-decomposed if a specified number of wells are allocated to each layer and there is no explicit consideration of vertical interaction. The decomposed model is of the form:

For each HSI $k$:

$$\text{Max } Z = \sum_{i \in I_k} (w_{sk}W_{ik}y_{ik} + w_{tk}W_{ik}z_{ik}) + \sum_{i \in I_a} W_{ik}y_{ik}$$ (3.39)

subject to

$$\sum_{j \in N_{ik}} x_{jk} \geq y_{ik} \quad \text{for each } i \in I_k$$ (3.40)

$$\sum_{j \in M_{ia}} x_{jk} \geq z_{ik} \quad \text{for each } i \in I_{dk}$$ (3.41)

$$y_{ik} + z_{ik} \leq 1 \quad \text{for each } i \in I_{dk}$$ (3.42)

$$x_{jk} = 0 \quad \text{for each } j \in J_{ik}$$ (3.43)

$$\sum_{j \in J_{zk}} x_{jk} = P_{zk} \quad \text{for each } z \in Z_k$$ (3.44)

$$\sum_{j \in J_{ik}} x_{jk} = P_k$$ (3.45)

$$y_{ik} = (0,1) \quad \text{for each } i \in I_k$$ (3.46)

$$z_{ik} = (0,1) \quad \text{for each } i \in I_{dk}$$ (3.47)

$$x_{jk} = (0,1) \quad \text{for each } j \in J_k$$ (3.48)

Variables retain definitions given for the general MWB model formulation. The parameter $P_k$ denotes the specified number of wells to be allocated to a given layer $k$. The general and layer-decomposed MWB model formulations can be solved with the branch-and-bound technique referenced in the preceding section.

3.3 Nodal Weights

Weights are used to quantify the value of potential sampling sites with regard to detection and exposure hazard. In addition to monitoring near the contamination source, monitoring at water supply wells in the regional vicinity of a waste facility is warranted because it can provide direct confirmation of a contamination problem at locations where exposure hazard is high. A "hydrogeologic term" quantifies relative detection value at nodes within the primary model domain. Weights for selected water supply wells are quantified by an "exposure term." For each candidate monitoring well site, $i$, nodal weights are calculated as
\[ W_i = (w_1)H_i / H(\text{max}) + (w_2)E_i / E(\text{max}) \]  

(3.49)

**Hydrogeologic Term:**

\[ H_i = \frac{1}{D_s D_{pi}} \]  

(3.50)

**Exposure Term:**

\[ E_i = P_{si} C_{hi} \]  

(3.51)

\[ C_{hi} = \frac{1}{D_s D_{pi}} \]  

(3.52)

where

- \( W_i \) = nodal weight
- \( w_1 \) = partitioning weight varied between zero and one
- \( w_2 = 1-w_1 \)
- \( H_i \) = value for hydrogeologic term (quantifies relative detection value)
- \( H(\text{max}) \) = reference value for hydrogeologic term (maximum value calculated for all nodes)
- \( E_i \) = value for exposure term (quantifies relative exposure hazard)
- \( E(\text{max}) \) = reference value for exposure term (maximum value calculated for all nodes)
- \( D_{si} \) = distance from candidate site to contaminant source
- \( D_{pi} \) = perpendicular distance from candidate site to advective envelope (explanation to follow in current section) (= \( D_p(\text{max}) \) if \( D_{pi} \) is undefined)
- \( P_{si} \) = population served by water supply well
- \( C_{hi} \) = contaminant hazard factor at water supply well

The hydrogeologic and exposure terms are normalized (divided) by the maximum values calculated for these terms for all candidate well sites. This procedure causes each term to vary within the same range, between zero and one. The values of the scaled hydrogeologic and exposure terms are independent of the units used for the distance parameters, \( D_{si} \) and \( D_{pi} \). Given two candidate monitoring well sites, one with a high detection term value and the other with a high exposure term value, the corresponding normalized weight values will be similar in magnitude, both near a value of one. This desired result cannot be achieved by scaling with a value alternative to the maximum, such as the mean.
The detection term is inversely proportional to each of two distance variables defining the proximity of a node to locations of high contamination susceptibility. Expressions alternative to (3.52), which utilize the distance parameters $D_{si}$ and $D_{pi}$, can also be used to define a contaminant hazard factor. One such alternative expression is $1/(D_{si} + D_{pi})$, in which the hazard factor is inversely proportional to the sum of the distance parameters. This condition is undesirable, however, because in situations where one distance parameter is large and the other small, the larger parameter may completely determine the value of the term as a whole. A smaller distance parameter may, in fact, signal the need for monitoring at the corresponding location and therefore should not be excluded in the derivation of the weight.

The hydrogeologic term which quantifies detection value is consistent with qualitative guidelines used in practice. For example, RCRA guidelines (U.S. Environmental Protection Agency, 1986) specify that the placement of downgradient detection wells must consider (1) the distance to the contaminant source and the direction of ground water flow, (2) the likelihood of intercepting potential pathways of contaminant migration, and (3) the characteristics of the contaminant source controlling the movement and distribution of contamination in the aquifer. Distance to the contaminant source is quantified by the variable $D_{si}$. The direction of ground water flow is considered in the definition of advective envelopes. In a general sense, the likelihood of intercepting potential contamination may be related to the distance between a node and high susceptibility areas. Finally, the characteristics of the contaminant source boundaries are considered in the definition of hydrogeologic outlets, from which advective envelopes are extended. In effect, the weighting methodology developed herein quantifies the qualitative guidelines listed above. Quantification facilitates a ranking of the relative value of a set of potential monitoring sites.

If ground water velocity varies significantly over a model domain, Equation (3.50) can be modified to

$$H_i = \frac{v_i}{D_a D_{pi}}$$

Equation 3.53 defines the detection term as directly proportional to velocity and inversely proportional to the distance parameters. Emphasizing variations in velocity is important because (1) high velocity areas often correspond to preferential contaminant migration pathways, and (2) the potential consequences of failing to detect migrating contaminant are greater in high velocity zones where an undetected plume can move relatively rapidly. If velocity is relatively uniform throughout a model domain, it may have little importance in quantifying detection value and can be set to the same value for all potential monitoring well sites. In this case, detection value is assessed solely from distance criteria, and Equation (3.50) can be employed.

Advective envelopes are constructed to delineate zones susceptible to becoming contaminated in the event of a release from the source. Envelope boundaries, originating from hydrogeologic outlets at contaminant source
boundaries, are constructed at right angles to hydraulic head contours (see figure 3.2). The example in figure 3.2 represents an advective envelope constructed for an upper interval in the zone of saturation for which the water table exerts the controlling influence on the movement of ground water and migrating contaminants. Similar advective envelopes can be constructed for deeper hydrostratigraphic intervals (HSI's). For deeper intervals, advective envelopes are extended laterally from areas of downward vertical flux, as interpreted from measurements, within facility boundaries. If vertical flux is directed downward, surface contaminants can travel into deeper HSI's and subsequently move horizontally beyond facility boundaries in response to hydraulic gradients characteristic of piezometric surfaces for lower elevation horizons.

Advective envelopes are not utilized as actual representations of the geometry of future contaminant plumes, nor as zones which encompass all possible contamination migration pathways. Local heterogeneities will inevitably lead to irregular migration pathways and plume geometries. However, in a regional scale analysis, it is not feasible to account for all such heterogeneities and their effects on contaminant migration. The advective envelope simply establishes a region of high contamination susceptibility based on an intuitive analysis of available hydrogeologic data. The analysis does not account for the effects of hydrodynamic dispersion which may result in contaminant spreading beyond advective envelope boundaries. However, nodes located outside such boundaries are treated as potential monitoring sites. The monitoring value of these nodes is weighed according to distance from the advective envelope, which is consistent with general tendency for progressive outward spreading. Establishing zones of high contamination susceptibility such as the advective envelope facilitates the computation of nodal weights which quantify the relative value of numerous potential sampling sites.

The exposure term expressed by Equations (3.51) and (3.52) is defined as a quantity directly proportional to (1) the population relying on a water supply well for consumption, and (2) the contaminant hazard factor, \( C_{hi} \). Ideally, the design of a ground water quality sampling plan would be based on reducing risk of injury, disease, or death resulting from human exposure to a potential environmental hazard in the subsurface (National Research Council 1983). However, determination of environmental risk, a procedure involving exposure hazard, dose-response evaluations, and risk characterization, is laborious, and the formulation of a ground water monitoring program based on reducing environmental risk is impractical. Risk can be defined as the probability of an adverse event times the consequence of the event. Determination of the probability of contamination at various points in uncontaminated three dimensional hydrogeologic settings is generally not feasible due to an absence of existing observations of chemical constituent concentrations at wells (supply wells are unaffected by the contaminant source prior to network design) and the complexity of hydrogeologic conditions, data uncertainty, and the related unfeasibility of extensive three dimensional numerical contaminant transport simulation.

The exposure term employed in this study is a proxy for environmental risk. In a general sense, the probability that a supply well is contaminated may be related to its position with respect to the contaminant source and likely zones of contaminant migration. The "consequence" of contamination of a water supply well may be related to the population relying on the well for consumption. Therefore, the two key notions of "probability" and "consequence," inherent in the definition of risk, are indirectly incorporated into the exposure term.
Depending on the goals of a particular study, it may be appropriate to give more emphasis to detection versus exposure (or vice-versa). Therefore, the detection and exposure terms are weighed according to the overall objective of the monitoring program. A program with a primary emphasis on early detection should include strategically located wells within the primary area of a model domain. Such a program should generally exclude water supply wells with low value as detection wells. This strategy can be implemented by choosing a high value for the weight $w_1$ (i.e., a value near one) and a low value for $w_2$. If, on the other hand, exposure hazard is of primary significance, water supply wells should be preferentially included in the monitoring program. This can be achieved by setting the weight $w_2$ to a value close to one and setting $w_1$ to a value near zero. The $w_1$ and $w_2$ weights are referred to as "partitioning weights."

The models do not explicitly define monitoring well construction details such as screen length and well diameter. These details require user judgment. Two- and four-inch (five- and ten-cm) diameter monitoring wells with five- to ten-foot (one- to three-m) screen intervals are commonly utilized in practice. One possibility would be to use similar diameters for all wells and to screen the monitoring wells throughout the depth of a hydrostratigraphic interval at a model defined location $(i,k)$. If smaller screen intervals are desired to provide more localized data, wells could be screened at consistent depths, for example, in the middle of an HSI.

3.4 Upgradient Monitoring

Upgradient (background) monitoring should be an important component of any ground water monitoring program (U.S. Environmental Protection Agency 1986). Ground water monitoring wells used for background sampling should be screened within deposits similar in character to those around downgradient monitoring wells (to ensure that only background conditions are registered), but away from potential contaminant migration pathways. There is some degree of qualitative judgment in the specification of the set of nodes among which background monitoring wells are to be allocated. The following guidelines are suggested to facilitate the proper selection of such nodes. (1) The nodes should not be within or adjacent to any advective envelope extended from the contaminant source. (2) The nodes should be located such that they cannot be intersected by an outward normal (i.e., perpendicular) line extending from an advective envelope. (3) The nodes should be located a distance of more than one grid unit from all contaminant source boundary nodes (i.e., nodes adjacent to the contaminant source boundary nodes should be excluded for upgradient monitoring). (4) The nodes should occupy positions with topographic elevations equal to or greater than the elevation at nearby contaminant source boundary nodes.

3.5 Temporal Changes in Monitoring Configuration

The network design models and associated weighting schemes are tailored to regional scale contaminant pollution problems, are applicable to layered aquifers, and emphasize the spatial (versus temporal) aspects of ground water quality monitoring. Two issues are important with regard to temporal monitoring: sampling frequency and timing of well siting. Sampling frequency is not considered within the model framework. In
current practice, regulators determine the frequency of sampling. For example, the California Central Coast Regional Water Quality Board specifies quarterly sampling for regulated facilities within its jurisdiction.

Initially, the models derive monitoring networks prior to the actual contamination of an aquifer. However, the models can also be used to choose new sampling locations as information becomes available on contaminant concentrations in an aquifer after a release from the source. This is accomplished solving the model again, setting the decision variable $x_{jk}$ equal to one at existing well sites which have registered contamination. The addition of this constraint would ensure the inclusion of these valuable well sites in subsequent monitoring configurations. These sites would also be used to constrain the orientation of advective envelopes, which would then be used to derive new nodal weights. This procedure would result in the preferential siting of wells near and downgradient of points of confirmed contamination. The procedure could be conducted at several points in time to track the growth of an evolving plume.
Fig. 3.1. Decay of Primary Covering Weight, $w_{si}$, as a Function of Distance from Contaminant Source, $D_{si}$; $D_f$ - Decay Factor.
Fig. 3.2. Hypothetical Advective Envelope; Rectangular Area - Contaminant Source; Thick Lines Extending from Source - Boundaries of Advective Envelope; Thin Lines - Hydraulic Head Contours; Elevations in Units of Length
4. CASE STUDY 1

4.1 Site Description and Background

Case study 1 involves a solid waste landfill located in Butler County, Ohio on the flood plain of the Great Miami River (figures 4.1 and 4.2). The municipal landfill, in operation from 1971 to 1985, has contaminated an underlying buried valley aquifer. A total of twenty monitoring wells have been located throughout the study site, including sixteen downgradient and four upgradient wells. The entire twenty-well network is termed the "original" network (figure 4.3). A subset of this twenty-well network has been used in recent years to detect contaminant migrating from the landfill (figure 4.3).

4.2 Hydrogeology

The buried valley aquifer consists of unconsolidated glaciofluvial deposits comprised predominantly of unconsolidated coarse sand and gravel. Silt and clay deposits are present in thin, laterally discontinuous lenses. Results of field tests reported by Hudak and Loaiciga (1991) indicate the aquifer is relatively homogeneous, with an average hydraulic conductivity of approximately $10^{-3}$ m/s. The glaciofluvial deposits range from six to twelve m in thickness and overlie shale bedrock. The regions to the immediate northwest and southeast of the study area (figure 4.2) are upland surfaces underlain by glacial till. Till deposits range from one to twelve 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 figures 4.1 and 4.2. 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 187 m above mean sea level. Depth to ground water characteristically ranges from approximately three to five m. The river is effluent (gaining) and in hydraulic connection with ground water in the underlying aquifer (figure 4.3).

4.3 Model Boundaries and Discretization

The grid of possible monitoring sites for the analytical model applications is shown in figure 4.4. The analytical model domain encompasses the region to the north of the river in figure 4.3. Though grid definition is somewhat arbitrary, the following guidelines can be used to determine an appropriate geometry for a given application. (1) A uniform orthogonal or triangular pattern is generally most effective for efficiently covering the region of primary emphasis. Orthogonal grids permit a direct comparison between the locations of sited wells and finite difference numerical model concentration estimates. However, where potential migration pathways are relatively narrow, triangular grids may be more effective because the pattern of alternating offset rows is more effective for early detection. (2) An inter-node spacing of approximately 100 to 500 feet (thirty to 150 m) is generally required for regional scale studies. A smaller spacing results in an excessive number of nodes needed to
cover the model area. Larger spacing may lead to inadequate detection capability near the contaminant source. (The study site meets the definition of a regional scale system given in Section 2.) (3) The total number of possible sites should be restricted to no more than a few hundred. This condition is necessary to limit computational requirements.

The node spacing clearly exceeds the scale of local heterogeneities which may be present within the aquifer. However, definition of contaminant concentration variations attributable to small-scale lithological variations is not the objective of a regional scale monitoring program. Instead, we seek to define a network which will detect contamination while collectively covering a broad portion of the plume.

Figure 4.4 illustrates two subsets of candidate well types: detection nodes, arranged in a regular geometric pattern, and existing supply wells, generally beyond the area occupied by detection nodes. Detection nodes are labeled according to position within the hydrogeologic setting. Landfill boundary nodes occupy the perimeter of the contaminant source and are located such that the separation between adjacent boundary nodes does not exceed one grid interval. Plume boundary nodes are located along the boundaries of the advective envelope. Upgradient nodes represent potential upgradient monitoring well sites. The locations of landfill boundary and plume boundary nodes are used in deriving the distance parameters, $D_{si}$ and $D_{pi}$.

In applying the analytical models to the Butler County case study, it was assumed that no information was available regarding the current presence of contaminants in the subsurface. The analytical models were applied to this case study as if it were a relatively new site for which an entire ground water quality monitoring network was to be derived. The number of wells located, $P_k$, in the (modeled) single layer system was set equal to the total number of wells in the original monitoring network (figure 4.3). This condition facilitates comparisons between analytical model-derived well configurations and the original network.

The procedure for calculating the nodal weights involves a relatively straightforward application of Equations (3.49) through (3.52). The user populations at each of the six supply wells was set equal to two. The exact number of current users for each of these wells is unknown. However, it is known that all of the wells are used for low-capacity domestic purposes. Under the assumption that the wells have similar user populations, the exact number of users input to the model is irrelevant, provided this value is identical for each well. This property results from the scaling procedure employed for the exposure term.

4.4 Numerical Modeling

A numerical ground water flow and mass transport model (MOC model, Konikow and Bredehoeft 1988) was used to generate a contaminant distribution for analytical model testing. Chloride concentrations were used to delineate the spatial extent of the plume. Numerical simulations of chloride transport provide conservative results because plumes defined by chloride are likely to be larger than plumes of other chemicals which may be retarded as a result of sorption or chemical processes. The numerical model was calibrated to hydraulic head and contaminant concentrations measured at pre-existing well sites (figure 4.3). Hudak and Loaiciga (1991) provide a detailed
description of field data collection, numerical modeling, calibration, and verification results. The verified model was used to estimate an equilibrium landfill-derived chloride concentration distribution. This distribution, attained after a period of about fifty years, was used as a basis for testing alternate monitoring network configurations. These results test the capability of monitoring well configurations to detect contaminant throughout the areal extent of the fully developed plume.

4.5 Network Configurations

4.5.1 Original Network. The original network, consisting of twenty wells (four designated for background monitoring) is shown in figure 4.5 along with the distribution of contamination predicted from numerical modeling. The well sites were selected for the purpose of detecting and defining future contamination beyond the waste facility boundaries (Woodward-Clyde and Canonie 1988). Qualitatively, the network has good detection capability due to a relatively small inter-well spacing near the contamination source. However, the configuration does not provide uniform coverage further downgradient and therefore lacks high characterization potential. The location of the four upgradient wells at nodes immediately adjacent to the source is undesirable because these points could become affected by contaminant spreading. This problem is avoided with analytical models by omitting nodes adjacent to the source from the set of potential upgradient monitoring sites.

The original network and contaminant field are depicted differently in figure 4.6 to facilitate assessment of network efficiency. In this figure, contaminated nodes coincident with or adjacent to sited wells are shaded. The shading convention can be used to visually assess the degree of separation between wells in a given network. Where inter-well separation is relatively large, there will be less overlap in shaded areas around wells, and a greater extent of the contaminant field will be shaded, or "covered." Assessment of network efficiencies is deferred to Section 4.6.

4.5.2 Basic Model Network. Given the series of nodal weights and associated well indices (i.e., 1, 2, 3,..., 220, representing 214 detection nodes and six existing supply wells), the specified number of wells to be located, and the partitioning in emphasis between detection and exposure, the optimal solution to the basic model was determined by solving the formulations presented in Equations (3.15) through (3.20). The nodes with the highest weights were obtained with sort functions in the S statistical package (Becker et al. 1988).

The basic model network (figure 4.7) was obtained for the partitioning weight combination \( w_1 = 1.0, w_2 = 0.0 \). For this combination of values, the emphasis is on detection (versus exposure). Sited wells are clustered around the source of contamination where nodal weights are generally highest. Virtually every node adjacent to a downgradient margin of the contaminant source is occupied by a well site. The basic model thus provides a configuration that is geared toward early release detection near the contaminant source. However, the configuration lacks monitoring sites throughout the downgradient area of the model domain that could be used for plume characterization.
4.5.3 MWB Model Networks. The layer-decomposed MWB model formulation, given by Equations (3.39) through (3.48), 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 UNIX). A driver program written in FORTRAN 77 was used to prepare an MPS-format data input file for the LINDO programming package (Hudak 1991). Depending on values for input parameters, most problems were solved in about twenty to thirty minutes for the applications discussed in this study. The branch-and-bound solution technique was required for the vast majority of the problems; the solution to the linear programming problem with relaxed integer constraints rarely generated a solution for which all model variables were integer.

The optimal MWB model solution is illustrated in figure 4.8. The solution corresponds to the parameter set \( (w_1=1.0, w_2=0.0, D_r=0.0, ss=0.5) \). As discussed in Section 3.2, a low decay factor preserves the difference in magnitude between the primary and secondary covering weights and facilitates some degree of well clustering, especially near the contaminant source. This condition is desired where plumes may be narrow. Hydrogeologic conditions that may lead to the occurrence of relatively narrow plumes include high groundwater velocities, and fairly homogeneous zones of high hydraulic conductivity. A high degree of heterogeneity at the macroscopic scale leads to a tendency for increased mechanical dispersion (Bear 1979). A value for the standard secondary covering weight equal to about one-half the value of the standard primary covering weight was determined optimal for the MWB model on the basis of empirical results (Hudak 1991). The MWB model-derived configuration exhibits a general pattern of larger inter-well separation with increased distance from the contaminant source. This pattern enhances the potential for both detection and characterization.

The MWB model was solved again for the weight combination \( (w_1=0.1, w_2=0.9) \), and other parameters as above. The weight \( w_1 \) was set to 0.1 instead of 0.0 to prevent the detection term from being driven to 0.0 for all nodes. In such a case, all six supply wells would be selected, but there would be an infinite number of possible choices for the fourteen remaining wells because these wells would have identical nodal weights, equal to 0.0. The choice of 0.1 is somewhat subjective; any number for \( w_1 \) between 0.0 and 0.5, exclusive, will give greater emphasis to exposure hazard (versus detection) and result in the preferential inclusion of water supply wells as monitoring sites. In the model solution, illustrated in figure 4.9, all six water supply wells are selected as monitoring wells. Comparison with the solution emphasizing detection (figure 4.8) indicates that the current solution (figure 4.9) is not obtained by simply shifting the outer-most detection wells in figure 4.8 to the six supply well sites. With fewer detection wells, it is advantageous to generate some degree of inter-well spreading, rather than crowding wells inward near the source. The full twenty-detection well solution in figure 4.8 cannot be obtained by retaining previous solutions with fewer wells and successively adding new wells.

4.6 Model Testing

4.6.1 Detection Efficiency. The numerical model-generated equilibrium contaminant distribution was
used to test analytical model solution efficiency. Detection efficiency is defined as

\[ E_f = \left( \frac{N_{wb}}{N_{wt}} \right) \times 100 \]  

(4.1)

where

\( E_f \) = detection efficiency (percent)
\( N_{wb} \) = number of wells (excluding upgradient wells) in a network with concentrations above background
\( N_{wt} \) = total number of wells (excluding upgradient wells) in a network

Analytical model solutions corresponding to \( w_1 = 1.0 \) and \( w_2 = 0.0 \) were used for detection efficiency assessment, and upgradient monitoring wells were excluded. The detection efficiency gives the fraction of the total number of monitoring wells in a network that would detect contamination according to numerical model predictions. The quantity \( E_f \) can be used to compare the detection characteristics of different networks. Networks with higher \( E_f \) values are superior in that they include a greater number of wells adequately positioned to detect contamination.

Figures 4.6 through 4.8 illustrate the relationship between various model-derived network configurations and nodes receiving contamination for the Butler County Landfill case study. All of the wells in each of the configurations detect contaminant because the contamination is widespread. As a result, detection efficiency cannot be used as a criterion for quantitatively distinguishing between the adequacy of alternate network configurations for this case study. Comparison of the locations of upgradient wells and nodes receiving contamination in figures 4.6 through 4.8 indicates that at least one upgradient well is contaminated in each of the original, basic model-determined, and MWB model-determined configurations. At least three upgradient wells are contaminated in each of the original and basic model networks. This condition is undesirable because upgradient wells are established for the purpose of monitoring uncontaminated background concentration levels. However, the concentration levels at the upgradient wells are low relative to the levels at the other contaminated nodes. Furthermore, the concentrations were generated by a numerical model which is, by definition, susceptible to inaccurate results (e.g., numerical dispersion, where a sharp concentration front is "smoothed out").

4.6.2 Characterization Efficiency. In addition to detection efficiency, an important issue in the assessment of sited well efficiency is plume characterization. A second technique was employed to address this issue. For characterization purposes, monitoring wells should be located far enough apart to maintain an adequate areal coverage over a potential contaminant plume. Characterization efficiency is assessed qualitatively by determining the percentage of a plume covered by the two-dimensional extent of sited wells. To facilitate assessment of total contaminant covered by wells in a given configuration, coverage is defined such that individual wells cover corresponding nodes and immediately adjacent (i.e., at one grid unit) nodes.

The original groundwater monitoring network covers only forty-four percent of the contamination predicted by numerical modeling for the Butler County study site (see figure 4.6, table 4.1). This relatively low characterization efficiency suggests that the network was not strategically designed to allow characterization and
monitoring throughout the region potentially contaminated by the landfill. The original network fully characterizes the portion of the aquifer near the source, with significant coverage overlap (i.e., nodes are covered by more than one monitoring well). This pattern is effective for release-detection, but ineffective for characterizing the downgradient extent of a plume. Wells sited at locations further from the source seem arbitrarily sited and discontinuously cover different parts of the zone of contamination (figure 4.6). There is not a uniform coverage of the contaminated region progressively outward from the source. For example, the south-central portion of the domain is covered, but the west-central and northwest parts of the contaminated region are generally uncovered.

The basic model-derived configuration does very poorly in characterizing, or covering the region of aquifer contaminated by the Butler County landfill (figure 4.7). Only twenty-three percent of contaminated nodes are covered by monitoring wells located by the basic model (table 4.1). This result reinforces the previously stated contention that the basic model is generally not suitable for deriving networks where characterization is important.

The MWB model-derived configuration out-performs the other configurations in covering the zone of contamination predicted by the numerical model. The configuration covers fifty-seven percent of the contaminated nodes (figure 4.8, table 4.1). This percentage, though well below 100, is significantly higher than the results obtained for the original and basic model configurations. The occurrence of a significant number of uncovered contaminated nodes (fourty-three percent) is attributed to the extensive area of contamination and the limited number of wells located. A summary of characterization efficiencies for each of the three configurations is listed in table 4.1.
TABLE 4.1
Summary of Monitoring Well Network Characterization Efficiencies, Butler County Landfill

<table>
<thead>
<tr>
<th></th>
<th>Original Network</th>
<th>Basic Model</th>
<th>MWB Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of Wells Located</td>
<td>20</td>
<td>20</td>
<td>20</td>
</tr>
<tr>
<td>Number of Contaminated Nodes</td>
<td>195</td>
<td>195</td>
<td>195</td>
</tr>
<tr>
<td>Number of Contaminated Nodes Covered</td>
<td>86</td>
<td>45</td>
<td>111</td>
</tr>
<tr>
<td>Characterization Efficiency (percent)</td>
<td>44</td>
<td>23</td>
<td>57</td>
</tr>
</tbody>
</table>
Fig. 4.1. Butler County, Ohio Landfill Study Area
Fig. 4.2. Schematic Geologic Cross-Section through Study Area
Fig. 4.3. Original Monitoring Well Locations (Circles and Dots), Existing Monitoring Well Locations (Circles), and Observed Steady-State Head Distribution; Elevations in Meters above Mean Sea Level (msl); Arrows Denote Boundaries of Advective Envelope
Fig. 4.4. Analytical Model Representation, Butler County Landfill; Small Dots - Potential Well Sites; Large Dots - Supply Well Sites; Triangles - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

Fig. 4.5. Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Butler County Landfill; Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)
Fig. 4.6. Contaminant Nodes Covered by Original Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

Fig. 4.7. Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)
Fig. 4.8. Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Butler County Landfill; Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)

Fig. 4.9. MWB Model-Determined Well Sites (Crosses and Large Dots), Butler County Landfill; $ss = 0.5; D_f = 0.0; w_1 = 0.1; w_2 = 0.9$; Squares - Landfill Boundary Nodes; Scale - Distance between Nodes along Rows = 375 Feet (114.3 Meters)
5. CASE STUDY 2

5.1 Site Description and Background

The Casmalia Resources hazardous waste facility is located in northern Santa Barbara County, approximately 1.6 miles (2.6 km) north of the unincorporated community of Casmalia (figure 5.1). The facility, originally permitted to dispose of hazardous (Class I) waste in 1972, comprises a total operating area of approximately one square kilometer. Hazardous and non-hazardous waste (liquid and solid) have been disposed at the facility during the seventeen-year period from 1972 to 1989. Solid waste has been disposed in six separate landfills, and liquid waste has been placed into a network of ponds and evaporation pads. A total of forty-three ponds and fifteen evaporative pads have been constructed. Landfills are generally located in the northern part of the facility; ponds occur predominantly to the south (figure 5.2).

Ponds and landfills were dug directly into underlying claystone deposits. Claystone is the predominant geological formation throughout the study area. In 1987, tests required by the California Regional Water Quality Control Board indicated widespread contamination of ground water beneath the site. Approximately twenty toxic chemicals were detected at levels exceeding California state drinking water standards. Ground water quality data reported by Woodward-Clyde (1987a; 1987b; 1989) and Woodward-Clyde and Canonie (1988) suggest the extent of off-site, facility-derived contamination may be limited to areas adjacent to facility boundaries, generally along the southern margin of the waste facility.

5.2 Hydrogeology

The Casmalia Resources facility is located south of the Santa Maria ground water basin, within a distinct drainage basin in the Casmalia Hills (figure 5.3). The Santa Maria Valley ground water basin lies north and east of the Casmalia Hills, about 2.5 miles (4.0 km) north of the Casmalia Resources facility. Consolidated Tertiary rocks form the Casmalia Hills, as well as the boundaries of the Santa Maria ground water basin. Aquifers within the Santa Maria Valley consist of Upper Tertiary- and Quaternary-aged unconsolidated sediments. The unconsolidated sequence attains a maximum thickness of about 3,000 feet (920 m). Woodward-Clyde (1987a; 1987b) and Woodward-Clyde and Canonie (1988) have provided extensive reports on the hydrogeology, ground water monitoring, and water quality status of the Casmalia study area.

Alluvium overlies the claystone bedrock along stream channels (intermittently) draining the area around the site. The alluvium consists of sandy clays to clayey sands that range in thickness from a few feet to about fifty feet (fifteen meters). The claystone bedrock unit is marine in origin and classified as the Todos Santos Claystone Member of the Tertiary, Sisquoc Formation. The bedrock unit can be divided into two distinct hydrostratigraphic units: an upper, weathered layer; and a lower, unweathered layer. Both weathered and unweathered claystone units are fractured. The weathered claystone is highly fractured, with spacing between fractures ranging from a few centimeters to about twenty centimeters (Woodward-Clyde 1987b). Fracture apertures range from about 0.06 to
0.25 millimeters in width. The unweathered claystone is lightly fractured. The weathered rock varies in thickness from about thirty to sixty feet (nine to eighteen meters). The unweathered claystone extends vertically for up to 2,000 feet (600 m) in depth to underlying shales of the Tertiary, Monterey Formation. Figure 5.4 illustrates water table contours in the vicinity of the facility. Facility boundaries correspond roughly with local topographic drainage divides (see figures 5.3 through 5.5). The water table configuration controls offsite migration in upper hydrostratigraphic intervals. The potential distribution of contamination in deep HSI's is determined by the location of areas of downward flux within facility boundaries (figure 5.6) and the piezometric surface for deeper intervals (figure 5.6).

5.3 Model Boundaries and Discretization

The continuous boundary encompassing the facility boundary in figure 5.3 defines the lateral extent of the primary domain for analytical modeling and the numerical ground water flow and mass transport model (MT3D) domain. Primary model boundaries are located, where possible, along natural hydrogeologic boundaries. Lateral boundaries on the east and west margins of the primary model domain coincide with inferred ground water divides and discharge areas. The west boundary is coincident with a north-south trending intermittent stream channel (Casmalia Canyon). The northern third of the eastern boundary similarly coincides with an ephemeral stream; the middle part of the boundary runs along an inferred local ground water divide; and the southern half of this boundary traces along a portion of Shuman Canyon, an inferred ground water discharge area. The southern boundary of the primary model domain coincides with the intermittently flowing Shuman Creek, an inferred regional ground water discharge area. The northern primary model boundary is located somewhat arbitrarily in that it does not coincide continuously with natural hydrogeologic boundaries. Ground water flows laterally from this boundary into the primary model area. Vertically, the primary model area was separated into three hydrostratigraphic intervals (HSI's): an upper unit extending from land surface to the weathered/unweathered claystone bedrock contact (HSI 1); an intermediate unit extending from the weathered/unweathered bedrock contact to forty feet (twelve meters) below the contact (HSI 2); and a deep unit beyond forty feet (twelve meters) below the contact (HSI 3). The upper HSI includes weathered claystone and alluvium. HSI's are schematically illustrated in figure 5.5.

The secondary model domain is the region outside the primary model boundaries in figure 5.3. This region includes a portion of the Santa Maria Valley ground water basin, flanking the northern margin of the Casmalia Hills. The water supply wells shown in figure 5.3 (excluding the site within the primary domain) represent the set of possible monitoring sites within the secondary domain. Supply well labels (figure 5.3) indicate well identification numbers, well type, and estimated population relying on the well for consumption. Four well types are included, corresponding to various water uses: irrigation, stock supply, domestic, and public supply. Irrigation and stock wells are not used for direct human consumption. However, if these wells became contaminated, pumped water could pose an indirect exposure hazard through irrigation or consumption by cattle.
It is inappropriate to assign a user-population of zero to these well types, as this would drive the value of the exposure term to zero (see Equations 3.51 and 3.52) and eliminate such wells from consideration for monitoring. As an alternative, these wells were assigned user populations equal to the minimum user population estimated for the wells that are used for direct consumption. In this case, the corresponding value is two. User population estimates were obtained from Santa Barbara County Environmental Health Services (B. Fonts, personal communication, 1991). Wells 6 and 21 serve high user populations relative to the other supply wells. Well 6 represents the "Casmite" well, serving the town of Casmalia; well 21 is owned by the Union Sugar processing facility.

Of the twenty-one supply wells illustrated in figure 5.3, seventeen are located along the margin of the Santa Maria Valley. In the unlikely event that facility-derived contaminant migrated to the wells, detection of such contaminant in one or more of these wells would mitigate potential exposure hazard and signal the need for measures to prevent further migration toward supply wells in the interior of the valley. There is a remote possibility that contaminant could migrate through deep hydrostratigraphic intervals to the northeast of the site, in the direction of these supply wells.

The number of wells to be located in each of the three analytical model HSI's was set equal to the corresponding number of monitoring wells in each of these units proposed by Woodward-Clyde and Canonie (1988) for off-site monitoring (hereafter, the "original" network). The original network consists of

Layer (HSI) 1:
31 total monitoring wells
6 upgradient monitoring wells

Layer (HSI) 2:
35 total monitoring wells
7 upgradient monitoring wells

Layer (HSI) 3:
17 total monitoring wells
2 upgradient monitoring wells

A triangular grid was used for domain discretization. In a triangular grid, adjacent rows are offset by half a grid unit. This property effectively mitigates the potential for a contaminant plume to migrate undetected between adjacent columns. The discrete analytical model domains for layers 1 through 3 are illustrated in figures 5.7 through 5.9. The grid spacing for upper intervals is based on the size of hydrogeologic outlets at the perimeter of the facility and computational considerations. The smallest hydrogeologic outlet, located at the southern end of the facility, is approximately 350 feet (107 m) wide (figure 5.4). Grid spacing for layers 1 and 2 is 350 feet (107 m), and rows are oriented approximately perpendicular to the mean direction of ground water flow inferred from
hydraulic head contours. The 350-foot (107 m) grid spacing results in discrete networks of 397 and 528 nodes, respectively, comprising the primary model domains for layers 1 and 2.

The distribution of candidate well sites in layers 1 and 2 comprises only the primary model area shown in figure 5.3. The supply wells in the secondary model domain (figure 5.3) and the single supply well within the primary area are not screened within either of the upper two HSI's. The wells were included in the analytical model representation for the lower HSI (layer 3). Supply wells in figure 5.3 can be subdivided into two groups, according to location with respect to the contaminant source. The four wells closest to the source are screened in deposits characteristic of the lower HSI. The other wells are located in unconsolidated deposits within the Santa Maria River Valley. These deposits do not correspond to the HSI's characteristic to the vicinity of the waste facility. As a result, the Santa Maria Valley wells cannot be assigned to analytical model layers on the basis of which HSI water is pumped from. However, for these wells to be contaminated by facility-derived pollutants, contaminant would have to travel along deep flow paths moving to the northeast of the facility, initially within HSI 3. Therefore, the Santa Maria Valley wells were assigned to HSI 3.

For many applications, the horizontal grid spacing may be appropriately increased for deeper HSI's. In many regional scale flow systems, the hydraulic head distribution at successively greater depth intervals tends to become increasingly more uniform. At deeper intervals, a plume may spread over greater areal distances because (1) contaminant has to travel downward prior to reaching greater depths, incurring some degree of spreading in the process, and (2) there may be fewer "troughs" in deeper hydraulic head distributions to constrain migration pathways. Wider plumes at deeper intervals would dictate the use of increased grid spacing in analytical model representation. As a guideline for determining the number of nodes in a given layer, the following equation can be used.

\[
P_k / N_k = C
\]  

where

- \( P_k \) = number of wells to be located in layer \( k \)
- \( N_k \) = total number of possible well sites in the layer
- \( C \) = constant value that does not vary significantly between layers (0 < \( C < 1 \))

By employing Equation (5.1), the fractions, \( C \), of wells sited in each layer are approximately equivalent. Use of this equation facilitates similar patterns and spatial relationships between sited wells in each layer. A low percentage of sited wells in a given layer could result in clustering near the source, whereas a higher percentage could result in a situation where the grid completely dictates the analytically-derived solution (i.e., wells are sited at virtually every grid point). From Equation (5.1),

\[
N_k = P_k / C
\]  

44
For the current application, $C$ values for layers 1, 2, and 3 are: 36/397, 42/528, and 17/179 (or .09, .08, and .09), respectively.

In some areas, under topographic highs, the water table is below layer 1 and is located within layers 2 or 3. Figure 5.7 shows voids with no nodes where layer 1 is dry, thereby precluding the siting of monitoring wells at these locations. Comparison between figures 5.7 and 5.8 indicates the presence of similar voids for layer 2, but the void areas are not as extensive. Layer 2 is deeper than layer 1 and, as a result, there are fewer areas where the water table is below the base of layer 2. In a few locations, both layers 1 and 2 are dry (i.e., the water table is below layer 2). This accounts for the void areas in figure 5.8.

The water table configuration and the hydrogeologic outlets it defines are important factors governing the potential distribution of contaminants in layers 1 and 2. As a result, the overall analytical model representations for layers 1 and 2 are fairly similar. HSI 3 differs from HSI's 1 and 2 with regard to the nature of the hydraulic head distribution controlling horizontal ground water flow (figure 5.6). In figure 5.6, advective envelopes have been constructed from areas of downward flux within the waste facility boundaries. The model representation for HSI 3 (water supply wells in figure 5.3 and detection nodes in figure 5.9) differs from those for layers 1 and 2 in several ways. (1) There are fewer nodes in the representation for HSI 3. Nodes in HSI 3 are located at every other node in the networks used for HSI's 1 and 2. To maintain the relationship quantified in Equations (5.1) and (5.2), it was necessary to reduce the total number of candidate nodes in HSI 3, due to the smaller number of total wells sited. As suggested previously, an exceedingly high number of candidate nodes could result in a clustering of sited wells near the source. Clustering in deeper HSI's is inappropriate, especially where the hydraulic head distribution is relatively uniform and, together with the distribution of areas of downward vertical flux within the source, dictate the possibility of relatively wide migrating plumes. (2) Advective envelopes are wider for HSI 3. This characteristic results from the relatively extensive areas of downward vertical flux within facility boundaries used in constructing the envelopes for HSI 3, and the more uniform distribution of hydraulic head characteristic to the deeper HSI 3. (3) There are no internal void areas within HSI 3. The water table does not occur below HSI 3. Therefore, there are no areas in which the HSI is dry. (4) The distribution of candidate nodes for HSI 3 includes existing supply well sites, most of which are situated along the margin of the Santa Maria Valley.

5.4 Numerical Modeling

Numerical simulation was conducted for the region within primary model boundaries only. Hydrogeologic data in areas beyond these boundaries is insufficient to permit accurate numerical modeling. MT3D (Zheng 1990) and MODFLOW (McDonald and Harbaugh 1988) were used to simulate the advection, dispersion, and diffusion, of dissolved bromide in the three-dimensional flow system. The areal locations of numerical model nodes correspond to the locations of nodes in every other row in the analytical model representation for the upper layers. The set of nodes comprises an orthogonal network (Hudak 1991).
The three-dimensional ground water flow and contaminant transport model was calibrated with an extensive hydrogeologic data set available for the case study (Hudak 1991). Several contaminant transport simulations were conducted for various simulation time periods and model solution schemes. Because of the extremely low ground water velocities characteristic to the hydrogeologic system, excessively long simulation time periods (up to 5,000 years) had to be employed to observe the evolution of migrating contaminant plumes. Simulations utilizing the mixed Lagrangian-Eulerian MOC-based schemes became computationally inefficient for longer time periods (i.e., several hundred years or more). Therefore, the more computationally efficient pure finite difference scheme was used for longer-term simulations. The conditions leading to potential inaccuracies in solutions derived with this technique, such as sharp concentration fronts in high-velocity systems, are not in evidence for the hydrogeologic setting under study.

From review of model output for numerous simulations, the maximum extent of contamination was observed to occur after approximately 1,000 years. Review of model output for 1,000- and 5,000-year simulations indicated no appreciable difference in overall plume geometry. The 1,000-year model predicted contaminant distribution was used for testing analytical model-derived monitoring well configuration solutions.

5.5 Network Configurations

The layer-decomposed basic and MWB models with upgradient and downgradient zones were applied to each of layers 1, 2, and 3. Layer-decomposed versions were utilized because these formulations are most efficiently solved when an exact number of wells to be located within each layer is specified. As stated previously, the numbers of wells to be allocated were obtained from the original network to facilitate quantitative comparisons between alternative configurations. The basic model was solved with the S statistical Package (Becker et al. 1988). The LINDO optimization package was used to solve the layer-decomposed MWB model formulations for case study 2. Problem solution times ranged from approximately twenty minutes to two hours, although most problems were solved within twenty to thirty minutes.

5.5.1 Layer 1.

5.5.1.1 Original Network. Figure 5.10 illustrates the locations of original wells sites and contaminated nodes. The configuration includes wells near hydrogeologic outlets, thereby providing detection capability near the source. However, the lack of monitoring points to the south and southeast of the source could result in contamination migrating undetected in these areas. In general, the configuration lacks a pattern of clustering near the source and progressive separation outward (figure 5.11). Note that the distribution of numerical model-generated contamination has been extrapolated to "offset" analytical model nodes in figure 5.11 and subsequent figures.

5.5.1.2 Basic Model Network. The basic model configuration for layer 1 is illustrated in figure 5.12. Monitoring wells are grouped near the contaminant source and along nodes within or bordering advective...
envelopes (compare figures 5.7 and 5.12). Upgradient monitoring wells are sited at nodes in the specified upgradient portion of the model domain and form a band parallel to the northwest facility boundary. Overall, the configuration has high detection potential, as monitoring wells are sited near potential contamination outlets, but little capability for plume characterization in the event of migration significantly beyond facility boundaries.

5.5.1.3 MWB Model Network. A decay factor, $D_f$, equal to 0.8 was used with the MWB model for layer 1. This value is near the maximum value of 1.0 for this parameter. The maximum value was used for layers 2 and 3. The hydrogeologic deposits comprising layer 1 (weathered claystone and alluvium) are more conductive than those in layers 2 and 3 and would, therefore, tend to dictate narrower plumes of migrating contaminant. The decay factor is correspondingly set to a slightly lower value. The MWB model solution for layer 1 is illustrated in figure 5.13. The configuration exhibits a pattern of well clustering near the source with increased spacing between wells sited progressively outward. Wells located further from the source cover high-susceptibility regions along the advective envelopes (compare figures 5.7 and 5.13). Qualitatively, this configuration has both detection and characterization potential. The upgradient wells (northwest of the facility) in the MWB model solution exhibit some degree of separation and characterize a larger portion of the upgradient region than do the upgradient wells sited by the basic model for layer 1.

5.5.2 Layer 2.

5.5.2.1 Original Network. The original network for layer 2 (figure 5.14) has detection capability along outlets at the southern boundary of the source, but lacks characterization potential downgradient (figure 5.15). Many of the wells located upgradient of the source, in low-susceptibility areas, would be more effective for detection and/or characterization if positioned downgradient of the source.

5.5.2.2 Basic Model Network. The thirty-five well basic model solution for layer 2 is illustrated in figure 5.16. The configuration is similar to the basic model configuration derived for layer 1. Recall that layer 2 contains a greater number of candidate nodes. This property accounts for some of the differences in the basic model network configurations for layers 1 and 2 (e.g., at the northeast and southeast margins of the site). Differences in basic model solutions for layers 1 and 2 are also due to the larger number of wells sited in layer 2 (thirty-five versus thirty-one). The overall similarity in model solutions for layers 1 and 2 is a result of the specification of the water table as the controlling head distribution governing the migration of contamination in both layers.

5.5.2.3 MWB Model Network. The MWB model solution for layer 2 (figure 5.17) exhibits the key characteristics of the MWB model configuration for layer 1 (figure 5.13). As with basic model solutions, differences in MWB model solutions for layers 1 and 2 can be attributed to larger numbers of candidate nodes and total wells located in layer 2.
5.5.3 Layer 3.

5.5.3.1 Original Network. Figure 5.18 illustrates the locations of monitoring wells in the original network for layer 3. As with original networks for the upper layers, the configuration has some detection capability provided by wells near the downgradient margins of the source. The absence of wells distributed further downgradient suggests a low characterization potential (figure 5.19).

5.5.3.2 Basic Model Network. Layer 3 includes both detection nodes and water supply wells as candidate monitoring sites. The basic model was solved with an emphasis on detection. Correspondingly, the detection term partitioning weight, \( w_1 \), was set equal to 1.0 and the exposure term weight, \( w_2 \), to 0.0. Although layer 3 contains fewer candidate nodes than layers 1 and 2, the basic model configuration for layer 3 (figure 5.20) is similar to basic model configurations for layers 1 and 2. This similarity is evident in the locations of sited wells near the contaminant source. The solution for layer 3 differs from those for the upper layers in that wells are not located away from the source along narrow zones. This difference is due to the more uniform areal distribution of hydraulic head governing transport in the lower HSI and the geometry of contaminant source boundary nodes. The absence of monitoring wells downgradient from the contaminant source in the basic model configuration for layer 3 suggests a low characterization potential.

5.5.3.3 MWB Model Networks. The MWB model configuration for layer 3 with an emphasis on detection (i.e., \( w_1=1.0, w_2=0.0 \)) is illustrated in figure 5.21. The configuration exhibits some degree of clustering near the contaminant source, with increased spreading outward. The distribution of wells differs markedly from the detection-emphasized basic model solution for the same layer (compare figures 5.20 and 5.21). The MWB model configuration characterizes a greater portion of the field of contamination susceptibility downgradient from the source.

The MWB Model was solved again for layer 3 with an emphasis on exposure hazard (\( w_1=0.0, w_2=1.0 \)). The constraints requiring upgradient monitoring were relaxed. As a result, the model sites the seventeen wells solely among the twenty-one water supply well sites. The seventeen-well model-derived configuration is illustrated in figure 5.22. The four supply wells not chosen for the monitoring network have low user populations and are relatively distant from the contaminant source. Note that some of the wells included in the model-derived solution are further from the contaminant source than are the excluded sites. The more distant sites were chosen due to higher associated user-populations. Both distance from source and user population are important factors in the determination of supply well monitoring sites.

5.6 Model Testing

5.6.1 Detection Efficiency.

5.6.1.1 Layer 1. The various monitoring well configurations and the distribution of numerical model-predicted contamination for layer 1 are illustrated in figures 5.11 through 5.13. All monitoring wells (excluding...
upgradient wells) detect contamination in each of the original, basic model, and MWB model configurations. Table 5.1 summarizes detection efficiencies for Casimlia Waste Facility layer 1. As in case study 1, the occurrence of contaminated upgradient nodes can be attributed to numerical model inaccuracies (Hudak 1991).

5.6.1.2 Layer 2. In the original network for layer 2 (figure 5.15), twenty-three out of twenty-eight non-upgradient detection wells coincide with contaminated nodes. The basic model solution (figure 5.16) out-performs the original network in detection efficiency. Only one well in the basic model configuration does not detect contamination. In the MWB model network for layer 2 (figure 5.17), twenty-three out of twenty-eight wells detect contamination. The MWB model network is thus equivalent to the original network in detection efficiency. However, the wells that do not detect contaminant in the MWB model network are more strategically located than the non-detecting wells in the original network (compare figures 5.15 and 5.17). The non-detecting MWB model-derived well sites are located beyond the leading edge of migrating contaminant plumes, where contamination would be detected with further plume growth. A summary of network configuration detection efficiencies for layer 2 is listed in table 5.2.

5.6.1.3 Layer 3. In the original network for layer 3 (figure 5.19), thirteen out of fifteen non-upgradient wells detect contaminant. In this case, the original network is detection-efficient because wells not detecting contaminant are located in areas that would become contaminated with further plume growth. The basic model outperforms the original network. All but one of the wells in the basic model network configuration detect contamination (figure 5.20). Four of the wells in the MWB model configuration for layer 3 do not detect contamination (figure 5.21). However, these wells are effectively located just beyond the downgradient margin of the extent of contamination to facilitate characterization with further contaminant migration. Table 5.3 summarizes detection efficiencies for the three configurations in layer 3.

5.6.2 Characterization Efficiency.

5.6.2.1 Layer 1. The original network covers forty-nine percent of the contaminated nodes in layer 1 (figure 5.11, table 5.4). This percentage is relatively low given the limited extent of numerical model-predicted contamination and the relatively large number of wells sited. The original network is out-performed by the basic model configuration in characterization efficiency. The basic model configuration covers fifty-seven percent of the distribution of contaminated nodes (figure 5.12, table 5.4). The basic model sites wells near the contaminant source at the expense of "lost coverage" downgradient. As established in previous applications, the extent of this zone of lost coverage downgradient of the source can be significant. However, in the current application, the basic model network still covers a greater percentage of contaminated nodes than does the original configuration. The MWB model configuration significantly outperforms the other configurations in total coverage (figure 5.13 and table 5.4). Wells sited by the MWB model characterize a large portion of the region of contamination predicted in layer 1. Eighty-five percent of the contaminated nodes are covered. Table 5.4 summarizes characterization efficiencies in layer 1.
5.6.2.2 Layer 2. The patterns of coverage associated with the various configurations in layer 1 are also evident in layer 2 (figures 5.15 through 5.17). The original configuration is again out-performed by both model-derived configurations in characterization efficiency. Table 5.5 summarizes characterization efficiencies for layer 2. The original configuration covers fifty-seven percent of contaminated nodes, the basic model covers sixty-two percent, and the MWB model covers eighty-one percent. Thus, the MWB model configuration characterization efficiency significantly exceeds the characterization efficiencies for the other two configurations.

5.6.2.3 Layer 3. The areal extent of numerical model-predicted contamination is not as large in layer 3 (figure 5.18) as in the upper layers. Contaminant must travel to greater depths to reach the lower layer and, once present, moves at very slow rates due to the low hydraulic conductivity characteristic of the unweathered claystone comprising this unit. Due, in part, to the limited extent of area contaminated, the percentages of nodes covered by various configurations in layer 3 are higher than the corresponding values for layers 1 and 2. Figures 5.19 through 5.21 illustrate monitoring well configurations and contaminated nodes in layer 3. The order of relative characterization efficiency between the configurations is identical to the order established for layers 1 and 2. In total coverage, or characterization efficiency, the MWB model configuration out performs the basic model configuration which, in turn, out-performs the original network (table 5.6).
### TABLE 5.1
Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 1)

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<th></th>
<th>Original Network</th>
<th>Basic Model</th>
<th>MWB Model</th>
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<tr>
<td>Number of Detection Wells</td>
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<td>25</td>
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<tr>
<td>Number of Wells Detecting Contaminant</td>
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<td>25</td>
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<tr>
<td>Detection Efficiency (percent)</td>
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### TABLE 5.2
Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 2)

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<th>MWB Model</th>
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<tbody>
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<td>28</td>
<td>28</td>
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<tr>
<td>Number of Wells Detecting Contaminant</td>
<td>23</td>
<td>27</td>
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<td>Detection Efficiency (percent)</td>
<td>82</td>
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<td>82</td>
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### TABLE 5.3
Summary of Monitoring Well Network Detection Efficiencies, Casmalia Waste Facility (Layer 3)

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<tr>
<td>Number of Wells Detecting Contaminant</td>
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<td>Detection Efficiency (percent)</td>
<td>87</td>
<td>93</td>
<td>73</td>
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### TABLE 5.4

Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 1)

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<th>MWB Model</th>
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<tr>
<td>Number of Wells Located</td>
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<td>Number of Contaminated Nodes Covered</td>
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<td>Characterization Efficiency (percent)</td>
<td>49</td>
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### TABLE 5.5

Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 2)

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<th>Original Network</th>
<th>Basic Model</th>
<th>MWB Model</th>
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<tr>
<td>Number of Wells Located</td>
<td>35</td>
<td>35</td>
<td>35</td>
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<tr>
<td>Number of Contaminated Nodes</td>
<td>115</td>
<td>115</td>
<td>115</td>
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<tr>
<td>Number of Contaminated Nodes Covered</td>
<td>66</td>
<td>71</td>
<td>105</td>
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<tr>
<td>Characterization Efficiency (percent)</td>
<td>57</td>
<td>62</td>
<td>81</td>
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### TABLE 5.6

Summary of Monitoring Well Network Characterization Efficiencies, Casmalia Waste Facility (Layer 3)

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<th>Original Network</th>
<th>Basic Model</th>
<th>MWB Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of Wells Located</td>
<td>17</td>
<td>17</td>
<td>17</td>
</tr>
<tr>
<td>Number of Contaminated Nodes</td>
<td>32</td>
<td>32</td>
<td>32</td>
</tr>
<tr>
<td>Number of Contaminated Nodes Covered</td>
<td>25</td>
<td>28</td>
<td>32</td>
</tr>
<tr>
<td>Characterization Efficiency (percent)</td>
<td>78</td>
<td>88</td>
<td>100</td>
</tr>
</tbody>
</table>
Fig. 5.1. Casmalia Study Area (Santa Barbara County, California); 1 Mile = 1.6 Kilometers
Fig. 5.2. Distribution of On-Site Waste Impoundments, Casmalia Waste Facility; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987a)
Fig. 5.3. Water Supply Well Locations (Large Dots); Labels Indicate Well Identification Number, Well Type (I: Irrigation, D: Domestic, P: Public, S: Stock) and Assigned User Population (in Parentheses) (U.S.G.S. Topographic Map - Casmalia, Guadalupe, Santa Maria, Orcutt (CA) 7.5 Minute Quadrangles, 1982; Elevations in Feet msl; 1 Foot = 0.3048 Meters)
Fig. 5.4. Water Table Contours (Feet msl), Advective Envelopes (Shaded Areas), and Location of Hydrogeologic Cross-Section A-A'; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987b)
Fig. 5.5. Hydrogeologic Cross-Section A-A'; 1 Foot = 0.3048 Meters (after Woodward-Clyde, 1987b)
Fig. 5.6. Inferred Distribution of Hydraulic Head in Layer 3 at 200 to 300 Feet msl Interval (Contours in Feet msl) and Advective Envelopes (Light Shaded Area) Constructed from Areas of Downward Flux (Dark Shaded Areas) within Facility Boundaries; 1 Foot = 0.3048 Meters
Fig. 5.7. Analytical Model Representation, Casmalia Waste Facility (Layer 1); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 350 Feet (106.7 Meters)

Fig. 5.8. Analytical Model Representation, Casmalia Waste Facility (Layer 2); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 350 Feet (106.7 Meters)
Fig. 5.9. Analytical Model Representation, Casmalia Waste Facility (Layer 3); Dots - Potential Well Sites; Triangles Enclosing Dots - Nodes at or within Advective Envelope Boundaries; Black-Filled Squares - Landfill Boundary Nodes; Squares Enclosing Dots - Upgradient Nodes; Scale - Distance between Nodes = 700 Feet (213.4 Meters)

Fig. 5.10. Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Circles - Upgradient Wells; Circles Enclosing Diamonds - Upgradient Wells Receiving Contamination; Circle Enclosing Cross Superimposed on Diamond - Two Original Wells (One Designated for Upgradient Monitoring) at Site Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)
Fig. 5.11. Contaminant Nodes Covered by Original Well Sites (Crosses), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)

Fig. 5.12. Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)
Fig. 5.13. Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 1); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)

Fig. 5.14. Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Casmalia Waste Facility (Layer 2); Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Circles - Upgradient Wells; Circles Enclosing Diamonds - Upgradient Wells Receiving Contamination; Circle Enclosing Cross - Two Original Wells (One Designated for Upgradient Monitoring) at Specified Site; Scale - Distance between Nodes = 350 Feet (106.7 Meters)
Fig. 5.15. Contaminant Nodes Covered by Original Well Sites (Crosses), Casmalia Waste Facility (Layer 2); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)

Fig. 5.16. Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 2); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)
Fig. 5.17. Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 2); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 350 Feet (106.7 Meters)

Fig. 5.18. Comparison between Original Well Sites (Crosses) and Numerical Model-Determined Sites Receiving Contamination (Diamonds), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Crosses Superimposed on Diamonds - Well Sites Receiving Contamination; 2 - Two Original Wells at Specified Site; Circle - Upgradient Well; Scale - Distance between Nodes = 700 Feet (213.4 Meters)
Fig. 5.19. Contaminant Nodes Covered by Original Well Sites (Crosses), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 700 Feet (213.4 Meters)

Fig. 5.20. Contaminant Nodes Covered by Basic Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Diamonds Enclosing Dots - Contaminant Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 700 Feet (213.4 Meters)
Fig. 5.21. Contaminant Nodes Covered by MWB Model-Determined Well Sites (Crosses), Casmalia Waste Facility (Layer 3); Squares - Landfill Boundary Nodes; Black-Filled Diamonds - Covered Contaminant Nodes; Crosses Superimposed on Black-Filled Diamonds - Well Sites Receiving Contamination; Scale - Distance between Nodes = 700 Feet (213.4 Meters)
Fig. 5.22. Basic Model-Determined Well Sites (Large Dots), Casmalia Waste Facility (Layer 3); \( w_1 = 0.0; w_2 = 1.0 \) (U.S.G.S. Topographic Map - Casmalia, Guadalupe, Santa Maria, Orcutt (CA) 7.5 Minute Quadrangles, 1982; Elevations in Feet msl; 1 Foot = 0.3048 Meters)
6. SUMMARY AND CONCLUSIONS

The basic and MWB optimization models developed in this study are applicable to regional scale ground water monitoring network design in multi-layered ground water flow systems. The initial step in applying either model is the discretization of the area potentially contaminated by a waste facility into a series of candidate well sites. Nodal weights, assigned to each of the candidate sites, are calculated from hydrogeologic (detection) and exposure hazard criteria. The weights quantify monitoring value at potential sites distributed throughout the model domain.

The basic model selects a specified number of candidate sites with the highest nodal weights, subject to the inclusion of upgradient monitoring sites. As suggested by the results of model applications described in this study, this selection strategy results in a monitoring configuration characterized by a clustering of wells around high susceptibility areas, generally near the contaminant source. The basic model is relatively easy to formulate and solve and effectively identifies points of high contamination susceptibility. This model is highly suitable for siting wells in monitoring programs emphasizing (1) optimal contaminant release detection potential near the contaminant source, or (2) mitigation of potential exposure hazard through the monitoring of existing water supply wells. In case (2), the basic model selects as monitoring sites the existing water supply wells with the highest exposure hazard, defined on the basis of user population and well proximity to high contamination susceptibility areas. The basic model is generally ineffective in generating solutions with contaminant characterization potential on a regional scale. Where characterization of the areal extent of potential contamination is an important issue, the MWB model is more appropriate.

The MWB model performs effectively with regard to both of the key issues of contaminant release detection and characterization efficiency. Configurations derived by this model exhibit, in general, a progressive increase in inter-well spacing away from the contaminant source. This pattern results in (1) a high potential for early detection near the source of contamination, and (2) an areally extensive downgradient network for subsequent plume characterization. The optimal configuration is obtained through a strategic variation of model "covering weights," within established criteria and according to the characteristics of the ground water flow system under study. The properties of detection and characterization are fundamentally important to the problem of ground water quality monitoring network design. A regional scale methodology, such as that encompassing the MWB model, which effectively addresses these properties can lead to early contaminant release detection and implementation of remedial action, and assessment of the distribution of ground water contamination and exposure hazard at water supply wells.

Monitoring networks derived from the methods developed in this study are more effective than networks designed from subjective criteria. For each of the applications discussed in the preceding sections, a composite network efficiency can be defined as the average of detection and characterization efficiencies. This value represents an index of overall performance of a monitoring network (table 6.1). Both analytical model configurations out perform original networks in composite efficiency for all applications, with the exception of the
Butler County case study. This exception is a result of the extremely low characterization efficiency associated with the basic model, caused by well clustering near the contaminant source and the widespread extent of numerical model predicted contamination. Composite efficiencies for each network configuration can be averaged across all applications, thereby generating an application-averaged composite efficiency (table 6.1). The results suggest that the MWB model is suited to the problem of network design at the regional scale to achieve the combined goals of plume detection and characterization.

The results of this work provide definitive evidence on the value of network design via integer programming at the regional scale. Techniques developed in this study are generally applicable to two or three dimensional ground water flow systems, with one or more hydrostratigraphic units or vertical well-siting intervals. To our knowledge, these are the first network design methodologies directly applicable to regional scale, multi-layered ground water flow systems. Potential users of the techniques developed in this study include federal, state, and local regulatory agencies, hazardous waste facility owners and operators, and private enterprises (e.g., hydrogeologic consulting firms) with interest in environmental problems.
### TABLE 6.1

Summary of Monitoring Well Network Composite Efficiencies

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