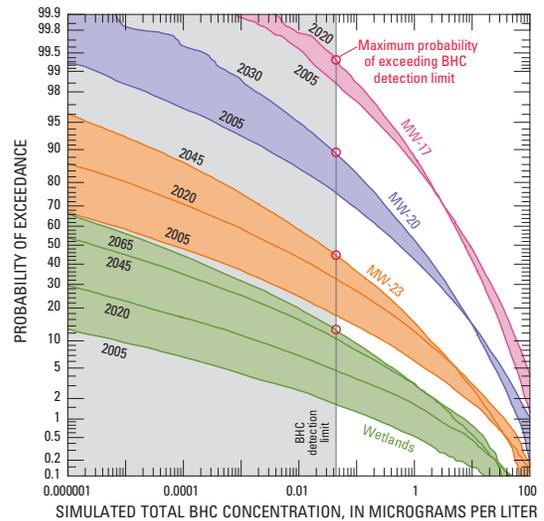
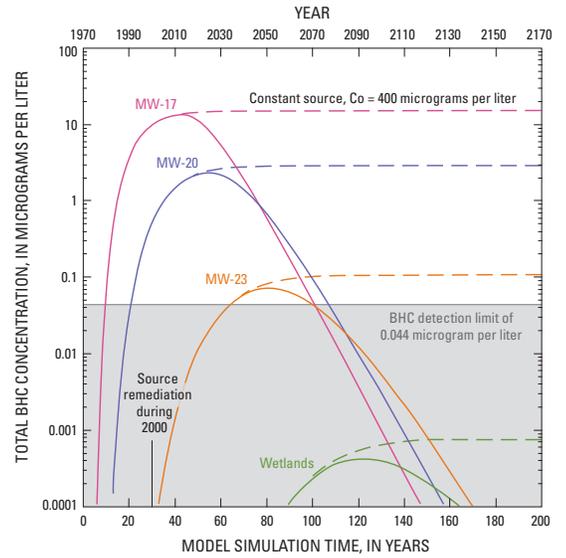
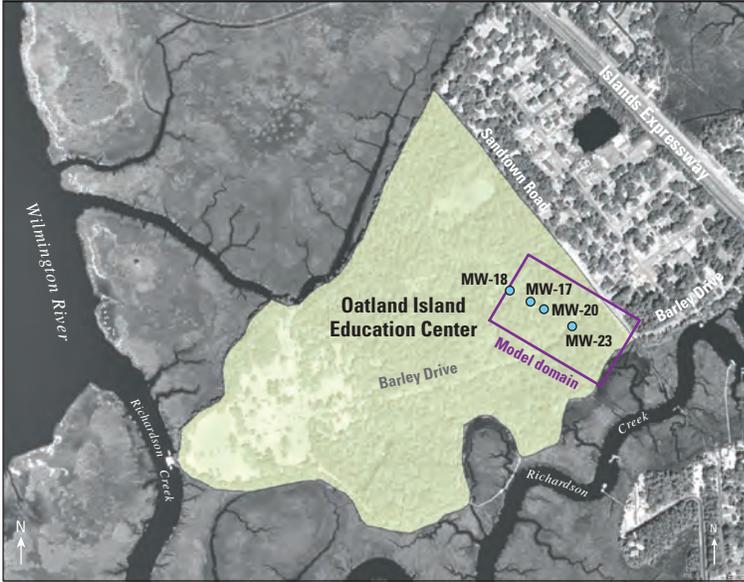


# Probabilistic Analysis of Pesticide Transport in Shallow Groundwater at the Oatland Island Education Center, Oatland Island, Georgia



*Front cover*

*Aerial photograph:* Oatland Island Education Center, Oatland Island, Chatham County, Georgia.  
1999 aerial photograph courtesy of U.S. Geological Survey through Terraserver USA

*Top photo:* Historic log cabin located at the former pesticide disposal area at Oatland Island Education Center

*Bottom photograph:* A Research Environmental Engineer with the Agency for Toxic Substances and Disease Registry collects information at one of the groundwater monitoring wells at the site

*Top graph:* Deterministic simulation results of total benzene hexachloride concentrations in groundwater for three key monitoring wells and the downgradient wetlands boundary

*Bottom graph:* Probabilistic simulation results expressed as probability of exceedance versus total benzene hexachloride concentration for three key monitoring wells and the downgradient wetlands boundary

# Probabilistic Analysis of Pesticide Transport in Shallow Groundwater at the Oatland Island Education Center, Oatland Island, Georgia

By Barbara A. Anderson, Morris L. Maslia, Jennifer L. Caparoso, and David Ausdemore

Prepared in coordination with the  
Centers for Disease Control and Prevention Office of Health and Safety

Agency for Toxic Substances and Disease Registry  
U.S. Department of Health and Human Services  
Atlanta, Georgia

September 2007

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## Foreword

The Agency for Toxic Substances and Disease Registry (ATSDR) is a public health agency in Atlanta, Georgia. ATSDR is part of the Centers for Disease Control and Prevention (CDC), within the U.S. Department of Health and Human Services. The agency's mission is to serve the public by using the best science, taking responsive public health actions, and providing trusted health information to prevent harmful exposures and diseases related to toxic substances.

ATSDR became involved with the Oatland Island site at the request of the CDC Office of Health and Safety (OHS) to evaluate potential public health impacts associated with pesticide contamination at the site. Based on available environmental data and exposure pathway information, ATSDR concluded in a 2005 health consultation report that the pesticide contamination does not pose a public health hazard to people who visit, live, or work at the site.

This report documents the continued collaboration between ATSDR and CDC OHS to examine the environmental fate and transport of pesticide contaminants in shallow groundwater at the site. Information about the migration patterns of the dissolved phase pesticide contaminant plume will be useful in guiding future environmental monitoring and remediation decisions for the site.



# Contents

Authors .....	ii
Foreword .....	iii
Abstract .....	1
Purpose and Scope .....	1
Background.....	2
Introduction.....	2
Site Geology and Hydrogeology.....	2
History of Contamination .....	2
Remediation Activities .....	5
Environmental Regulatory and Public Health Framework .....	5
Contaminant Properties and Health Effects.....	7
Methods.....	7
Modeling Approach.....	7
Assumptions .....	8
Reference Contaminant Selection.....	8
Model Input Parameters and Source Definition for Deterministic Simulations.....	9
Model Calibration Using Plume-Matching .....	9
Model Input Parameters for Probabilistic Simulations .....	11
Results .....	13
Model Calibration.....	13
Deterministic Simulations for an Eliminated Source .....	13
Deterministic Simulations for a Constant Source .....	16
Probabilistic Model Simulations .....	16
Probability Density Functions for Model Variants.....	16
Probabilistic Results.....	16
Probabilistic Type Curves for Each Site Location.....	16
Monitoring Well MW-23 .....	16
Wetlands Boundary.....	20
Summary Matrix for Selected Compliance Goals .....	20
Discussion.....	20
Recommendations .....	22
Limitations of Models and Analyses.....	22
Summary and Conclusions.....	23
References .....	24
Appendix A. Selected Illustrations from Previous Reports.....	27
Appendix B. Summary of Research on Media and Contaminant Characteristics .....	33
Appendix C. Calculating Site-Specific Aquifer Characteristics Using Tidal Information.....	37
Appendix D. Probabilistic (Monte Carlo) Simulations.....	41

## Figures

1–2.	Maps showing—	
1.	Study area, Oatland Island Education Center, Oatland Island, Chatham County, Georgia.....	3
2.	Study area and model domain, Oatland Island Education Center, Oatland Island, Georgia .....	4
3–13	Graphs showing—	
3.	Benzene hexachloride concentrations in groundwater measured at monitoring wells MW-18, MW-17, and MW-20 for site sampling events, 2000–2005.....	5
4.	Comparison of measured concentrations and deterministic simulation results for total benzene hexachloride concentrations in groundwater for 2001, 2003, and 2005 calibration targets.....	13
5.	Calibration targets superimposed on the spatial distribution of total benzene hexachloride concentrations in groundwater for 2003 (model simulation time = 33 years) .....	14
6.	Deterministic simulation results of total benzene hexachloride concentrations in groundwater for an eliminated source scenario during (A) 1975, (B) 1990, (C) 2000, (D) 2020, (E) 2050, and (F) 2080.....	15
7.	Deterministic simulation results of total benzene hexachloride concentrations in groundwater versus distance from the source along the contaminant plume centerline during 2000, 2020, 2050, and 2070.....	14
8.	Deterministic simulation results of total benzene hexachloride concentrations in groundwater for an eliminated source scenario during (A) 2000, (B) 2050, and (C) 2080 and 2090, and for a constant source scenario during (D) 2000, (E) 2050, and (F) 2090 .....	17
9.	Deterministic simulation results of total benzene hexachloride concentrations in groundwater over time for an eliminated source scenario and a constant source scenario.....	18
10.	Darcy velocity (specific discharge) values generated for probabilistic modeling: (A) frequency distribution (histogram) and (B) probability density.....	18
11.	Probabilistic simulation results expressed as probability of exceedance versus simulated total benzene hexachloride concentration for four site locations of interest.....	19
12.	Probabilistic simulation results expressed as probability of exceeding the benzene hexachloride detection limit of 0.044 microgram per liter versus simulation time for the eliminated source scenario at four site locations of interest .....	19
13.	Probabilistic model results at MW-23 for time periods (A) before and (B) after 2045, when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter .....	19
14.	Probabilistic model results at the wetlands for time periods (A) before and (B) after 2065, when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter .....	21

A1.	Map showing geologic cross-section locations and groundwater potentiometric surface map, March 2003.....	28
A2.	Geologic cross-section A–A' .....	29
A3.	Geologic cross-section B–B' .....	30
A4.	Map showing Area A groundwater quality and potentiometric surface, March 2003 .....	31
C1.	Map showing tidal study reference points, Oatland Island Education Center, Oatland Island, Chatham County, Georgia.....	39
C2.	Graphs showing tidal data and monitoring well measurements for Oatland Island, Georgia: (A) Fort Pulaski NOAA station ocean tide and Richardson Creek tide measurements and (B) water level measurements for Area B monitoring wells.....	40
D1–D4.	Graphs showing probability densities generated using 10,000 Monte Carlo realizations—	
D1.	(A) Darcy velocity, in feet per year, and (B) retardation coefficient (unitless) for benzene hexachloride.....	43
D2.	(A) longitudinal and (B) transverse dispersion coefficients, in feet squared per year .....	43
D3.	(A) infiltration (recharge) to the surficial aquifer, in feet per year, and (B) aquifer porosity (unitless).....	44
D4.	(A) aquifer thickness, in feet, and (B) standard deviation of the source width ( $\sigma_w$ ), in feet.....	44
D5–D8.	Graphs showing model results from probabilistic analysis at MW-20 during 2003 expressed as a—	
D5.	Frequency distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values .....	45
D6.	Probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values .....	45
D7.	Cumulative probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values.....	46
D8.	Complementary cumulative probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values.....	46
D9–D16.	Graphs showing probabilistic model results when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride detection limit of 0.044 microgram per liter—	
D9.	At MW-17 for time periods before 2020 .....	47
D10.	At MW-17 for time periods after 2020.....	47
D11.	At MW-20 for time periods before 2030 .....	47
D12.	At MW-20 for time periods after 2030.....	47
D13.	At MW-23 for time periods before 2045 .....	48
D14.	At MW-23 for time periods after 2045.....	48
D15.	At the wetlands for time periods before 2065 .....	48
D16.	At the wetlands for time periods after 2065.....	48

## Tables

1. Reference values for regulatory action and health guidance for benzene hexachloride in groundwater, Oatland Island, Georgia .....	6
2. Calibrated parameter values used to simulate the fate and transport of total benzene hexachloride in groundwater for deterministic analysis using ACTS software, Oatland Island, Georgia .....	10
3. Contaminant source characterization scenarios.....	11
4. Transport model calibration targets selected from available benzene hexachloride groundwater monitoring results, Oatland Island, Georgia .....	11
5. Parameter values used to simulate the fate and transport of total benzene hexachloride in groundwater for a probabilistic (Monte Carlo) analysis, Oatland Island, Georgia .....	12
6. Summary matrix of the maximum probability of benzene hexachloride in groundwater exceeding various regulatory compliance goals at selected site locations, Oatland Island, Georgia .....	21

## Conversion Factors

<b>Multiply</b>	<b>By</b>	<b>To obtain</b>
<b>Length</b>		
inch	2.54	centimeter (cm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
<b>Area</b>		
acre	4,047	square meter (m <sup>2</sup> )
square foot (ft <sup>2</sup> )	929.0	square centimeter (cm <sup>2</sup> )
square foot (ft <sup>2</sup> )	0.09290	square meter (m <sup>2</sup> )
<b>Volume</b>		
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m <sup>3</sup> )
<b>Flow rate</b>		
foot per day (ft/d)	0.3048	meter per day (m/d)
foot per year (ft/yr)	0.3048	meter per year (m/yr)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
<b>Mass</b>		
ton, short (2,000 lb)	0.9072	megagram (Mg)
<b>Density</b>		
pound per cubic foot (lb/ft <sup>3</sup> )	0.01602	gram per cubic centimeter (g/cm <sup>3</sup> )
<b>Hydraulic conductivity</b>		
foot per day (ft/d)	0.3048	meter per day (m/d)
<b>Hydraulic gradient</b>		
foot per mile (ft/mi)	0.1894	meter per kilometer (m/km)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$^{\circ}\text{F}=(1.8\times^{\circ}\text{C})+32$$

## Acronyms and Abbreviations

Definition of acronyms and abbreviations used throughout this report are listed below:

<b>Acronym or abbreviation</b>	<b>Definition</b>
ACTS	Analytical contaminant transport analysis system
ATSDR	Agency for Toxic Substances and Disease Registry
bgs	below ground surface
BHC	benzene hexachloride (also known as HCH, hexachlorocyclohexane)
<i>C</i>	concentration
ca.	calculated
$C_o$	initial concentration
CDC	Centers for Disease Control and Prevention
$D_x$	longitudinal dispersion coefficient
$D_y$	lateral dispersion coefficient
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DNAPL	dense non-aqueous phase liquid
$f_{oc}$	fraction of organic carbon in soil
ft	foot (feet)
Georgia EPD	Georgia Environmental Protection Division
gal	gallon
$g/cm^3$	gram per cubic centimeter
HCH	hexachlorocyclohexane (also known as BHC, benzene hexachloride)
hr	hour
HSR	hazardous site response

$i$	hydraulic gradient
$I$	infiltration
$K$	hydraulic conductivity
$K_d$	soil distribution coefficient or soil-water partition coefficient
$K_{oc}$	soil organic carbon partition coefficient
$K_{ow}$	octanol-water partition coefficient
MWH	Montgomery Watson Harza Americas, Inc.
MCL	maximum contaminant level
mg/L	milligram per liter
MW	monitoring well
OHS	Office of Health and Safety (Centers for Disease Control and Prevention)
$R$	retardation factor
RRS	risk reduction standard
TDL	Technical Development Laboratories, operated by the Office of Malaria Control of the Communicable Disease Center (now known as the Centers for Disease Control and Prevention)
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
$V_d$	specific discharge (Darcy velocity)
$V$	groundwater velocity (interstitial or pore velocity)
$\alpha_L$	longitudinal dispersivity
$\alpha_T$	transverse dispersivity
$\alpha_L/\alpha_T$	ratio of dispersivities
$\theta$	soil porosity
$\rho_b$	bulk density of soil
$\sigma_w$	standard deviation of contaminant source width, assuming a Gaussian distribution
$\mu\text{g/L}$	microgram per liter

Use of trade names and commercial sources is for identification only and does not imply endorsement by the Agency for Toxic Substances and Disease Registry, the Centers for Disease Control and Prevention, or the U.S. Department of Health and Human Services.



# Probabilistic Analysis of Pesticide Transport in Shallow Groundwater at the Oatland Island Education Center, Oatland Island, Georgia

By Barbara A. Anderson,<sup>1</sup> Morris L. Maslia,<sup>1</sup> Jennifer L. Caparoso,<sup>2</sup> and David Ausdemore<sup>2</sup>

## Abstract

The Agency for Toxic Substances and Disease Registry (ATSDR) used a publicly available analytical modeling software application, the analytical contaminant transport analysis system (ACTS), to examine the fate and transport of organochlorine pesticides in shallow groundwater at the Oatland Island Education Center, Oatland Island, Georgia. Specific objectives included (1) estimating the probability of impacting coastal wetlands located 800 feet downgradient of the pesticide source area, and (2) developing reference tools (probabilistic type curves) for evaluating future groundwater monitoring results at key site monitoring wells.

The groundwater contaminant plume consists of four different isomers of benzene hexachloride (BHC). Several other organochlorine pesticides, including aldrin and dieldrin, have been detected at the former source area but not in the downgradient portions of the contaminant plume. ATSDR selected total BHC as a reference contaminant for fate and transport model simulations. Model results are considered representative of the four BHC isomers at the site.

Based on groundwater monitoring and field data for the site, ATSDR selected an analytical model to analyze two-dimensional, saturated-zone groundwater contaminant fate and transport. The modeling incorporated probabilistic techniques to account for groundwater flow and contaminant transport parameter variability and uncertainty. Model input parameters were derived from site-specific field data, U.S. Geological Survey data for the surrounding area, and literature research.

Model calibration consisted of adjusting input parameter values iteratively to match model simulation results with actual contaminant concentrations measured over time in selected wells. Probabilistic model results for each specific site location and time period were developed using a two-stage Monte Carlo procedure that generated 10,000 realizations of eight model parameter variants.

Deterministic simulations using calibrated, single value input parameters indicate the contaminant plume will not impact the wetlands. Probabilistic simulations using 10,000 realizations for eight different input parameters indicate that the probability of exceeding the detection limit (0.044 microgram per liter) of total BHC in groundwater at the wetlands boundary increases from 1 percent (%) during 2005 to a maximum of 13% during 2065. This represents an 87% confidence level that the wetlands will not be impacted in the future by pesticide migration from the site.

At MW-23, a monitoring well located along the centerline of the plume, halfway between the former source area and the wetlands, model results indicate a 12% to 45% probability that total BHC will be detected at levels above the detection limit during 2000–2045. However, BHC was not detected at MW-23 in the eight sampling events conducted during 2000–2005. Long-term monitoring at MW-23—and at other key monitoring wells at the site—is a practical approach that enables evaluation of actual contaminant plume migration, comparison with model predictions, and timely decision-making for the site that will be protective of the wetlands.

ATSDR used the model simulation results to develop a series of probabilistic type curves for four different site locations—three downgradient monitoring wells and the downgradient wetlands boundary—along the centerline of the contaminant plume. These probabilistic type curves can be used to evaluate future groundwater monitoring results and to guide environmental and regulatory decision-making for the site.

## Purpose and Scope

The purpose of this project was to provide a reliable estimate of the long-term migration patterns of pesticide contamination in shallow groundwater at the Oatland Island Education Center, Oatland Island, Georgia. The source of the pesticide contamination in soil at the site was eliminated during 2000 (Montgomery Watson Harza Americas Inc. [MWH] 2001). No current public health issues are associated with the pesticide-contaminated groundwater at the site, primarily because shallow groundwater in the area is not used for

<sup>1</sup> Agency for Toxic Substances and Disease Registry, Atlanta, Georgia.

<sup>2</sup> Centers for Disease Control and Prevention, Atlanta, Georgia.

drinking water (ATSDR 2005a). However, future southward migration of the contaminant plume into the coastal wetlands along Richardson Creek is a potential ecological concern. Additionally, the pesticide concentrations in groundwater at the site exceed applicable state environmental regulatory levels.

Specific objectives of this work include: (1) estimating the probability of impacting coastal wetlands located 800 feet (ft) downgradient of the pesticide source area, and (2) developing reference tools (probabilistic type curves) for evaluating future groundwater monitoring results at key site monitoring wells.

## Background

### Introduction

The Oatland Island Education Center is located immediately east of Savannah, Georgia (Figure 1). The Center is owned and operated by the Savannah-Chatham County Public Schools as an environmental education facility. The 173-acre facility contains several buildings, wildlife enclosures, and trails (S&ME 1999a).

The Communicable Disease Center (now known as the Centers for Disease Control and Prevention [CDC]) and its predecessor agency, the Office of Malaria Control on War Areas, operated the Technical Development Laboratories (TDL) on the site during 1943–1973. In 1974, the U.S. Government deeded the property to the Savannah-Chatham Board of Public Education with the stipulation that the property be used for educational purposes for a period of 30 years.

In 1998, school officials discovered a map from 1973 that indicated the location of two onsite disposal areas labeled “Insecticide Burial Area” and “Radioactive Burial Area” (S&ME 1999b). The Insecticide Burial Area, designated as Area A, is the focus of this report (Figure 1). The Radioactive Burial Area, designated as Area B, has been remediated, and no further investigative or corrective actions are required (MWH 2001).

### Site Geology and Hydrogeology

Oatland Island is one of a number of islands along the eastern coast of Georgia, within the Lower Coastal Plain Province of Georgia. Layered sediments of sand, clay, and limestone in this province form a surficial aquifer (typically 65 ft thick in the vicinity of Oatland Island), the Upper and Lower Brunswick aquifers (occurring at depths of 88–340 ft below ground surface [bgs]), the Floridan aquifer system (occurring at depths of 110–530 ft bgs), and several other, deeper aquifer systems (Clarke et al. 1990). At Oatland Island and adjacent Skidaway Island (Figure 1), the surficial aquifer is divided into an upper unconfined zone and a lower semi-confined zone separated by a clay layer (MWH 2003, Clarke et al. 1990).

Oatland Island is surrounded by tidal marshes, and its topography is generally flat. Soil-boring and well-installation

logs from the site characterize subsurface soils as fine sand with some silt from ground surface to 14–23 ft bgs (MWH 2003, S&ME 1999b). Grain-size analyses of samples from these shallow soils indicate greater than 90 percent (%) fine sand (S&ME 1999b). Some laterally discontinuous lenses of clay and silt also were documented in this unit (MWH 2003). The fine sands are underlain by 7 to 8 ft of clay followed by a layer of fine to coarse sand (MWH 2003). Geologic cross-section diagrams from environmental investigations at the site are included in Appendix A (Figures A2 and A3).

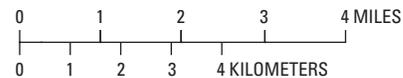
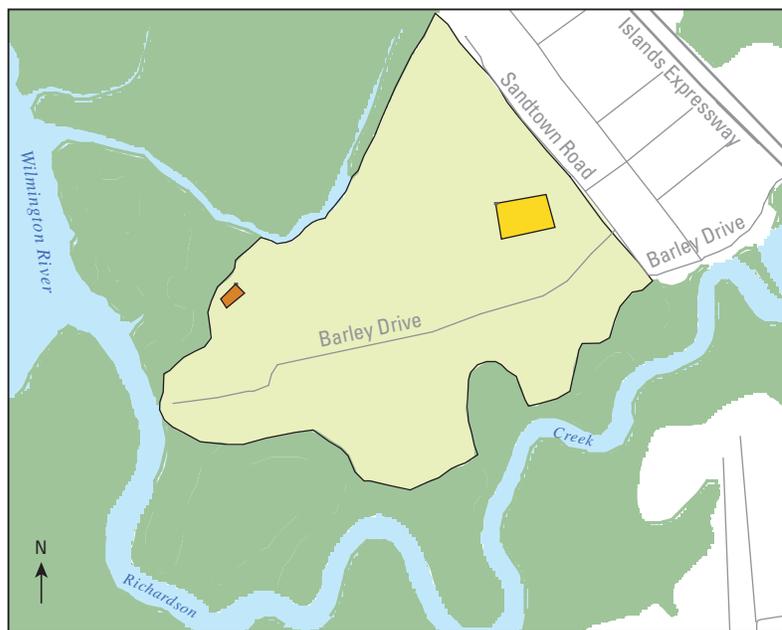
During site sampling events, groundwater was typically located at 5–9 ft bgs. Numerous groundwater monitoring events during 2000–2005 confirmed that the groundwater flow direction in the study area is to the east, as shown by the potentiometric surface map in Figure A4 (MWH 2003, MWH 2005).

### History of Contamination

In 1998, Savannah-Chatham school officials discovered a map dated July 1973 depicting an area where insecticides were disposed at the site. Follow-up interviews with former TDL employees confirmed that for a period of time before 1974, insecticides were buried at Area A in 1-gallon (gal), 5-gal, and 55-gal metal containers, cardboard boxes, and plastic bags (S&ME 1999b). Based on this information, CDC initiated environmental investigations by contracted consultants to delineate the nature and extent of soil and groundwater contamination at Area A (MWH 2001, S&ME 1999b). Site investigations included soil and groundwater sampling and installation of a network of permanent groundwater monitoring wells. Sampling results confirmed the presence of a number of organochlorine pesticides in the soil and groundwater at Area A:

- alpha-, beta-, delta-, and gamma-isomers of benzene hexachloride (BHC);
- aldrin and dieldrin;
- endosulfan; and
- dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethane (DDD), and dichlorodiphenyldichloroethylene (DDE).

The aldrin, dieldrin, endosulfan, and DDT-related compounds were detected in groundwater primarily within the source area. The groundwater contaminant plume extending downgradient from the source area consists of the four BHC isomers. The BHC groundwater contamination is limited to the unconfined zone of the surficial groundwater aquifer that is approximately 14 ft thick, with a confining clay layer at its base (Figures A1, A2, A3). The maximum concentration of total BHC in groundwater during 2000–2005 monitoring events was 5.38 micrograms per liter ( $\mu\text{g/L}$ ) at monitoring well MW-17 during November 2001 (Figures 2 and 3). Corresponding individual isomer concentrations ranged from less than 0.5  $\mu\text{g/L}$  for gamma-BHC to 3.6  $\mu\text{g/L}$  for delta-BHC (MWH 2002a, MWH 2002b, MWH 2005).

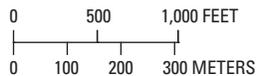


**EXPLANATION**

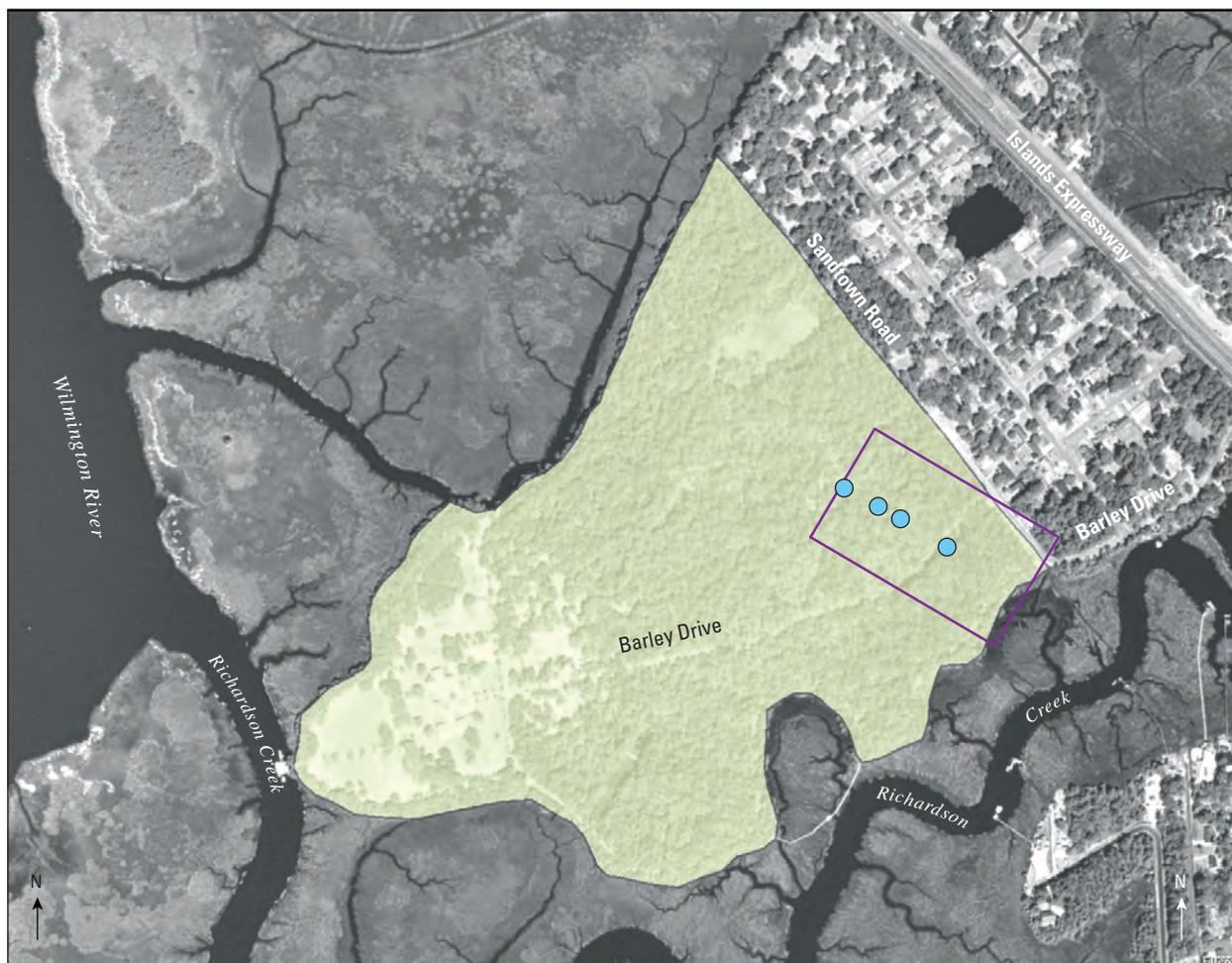
- Oatland Island Education Center
- Area A
- Area B
- Wetland
- Water



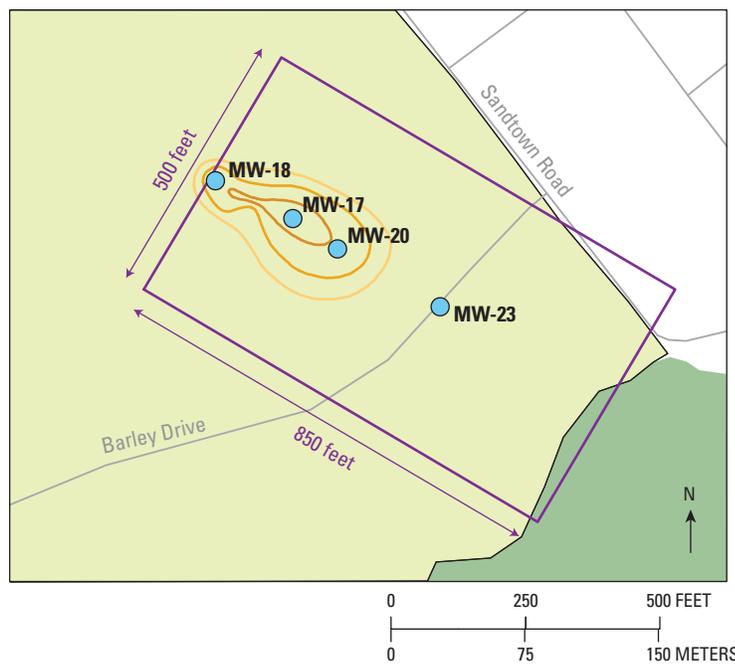
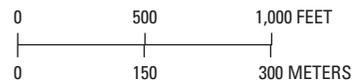
Base modified from Montgomery Watson Harza, 2005  
Wetlands boundaries based on  
National Wetlands Inventory data  
from U.S. Fish & Wildlife Service, 2004



**Figure 1.** Study area, Oatland Island Education Center, Oatland Island, Chatham County, Georgia.

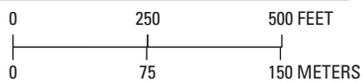


1999 aerial photograph courtesy of U.S. Geological Survey through Terraserver USA



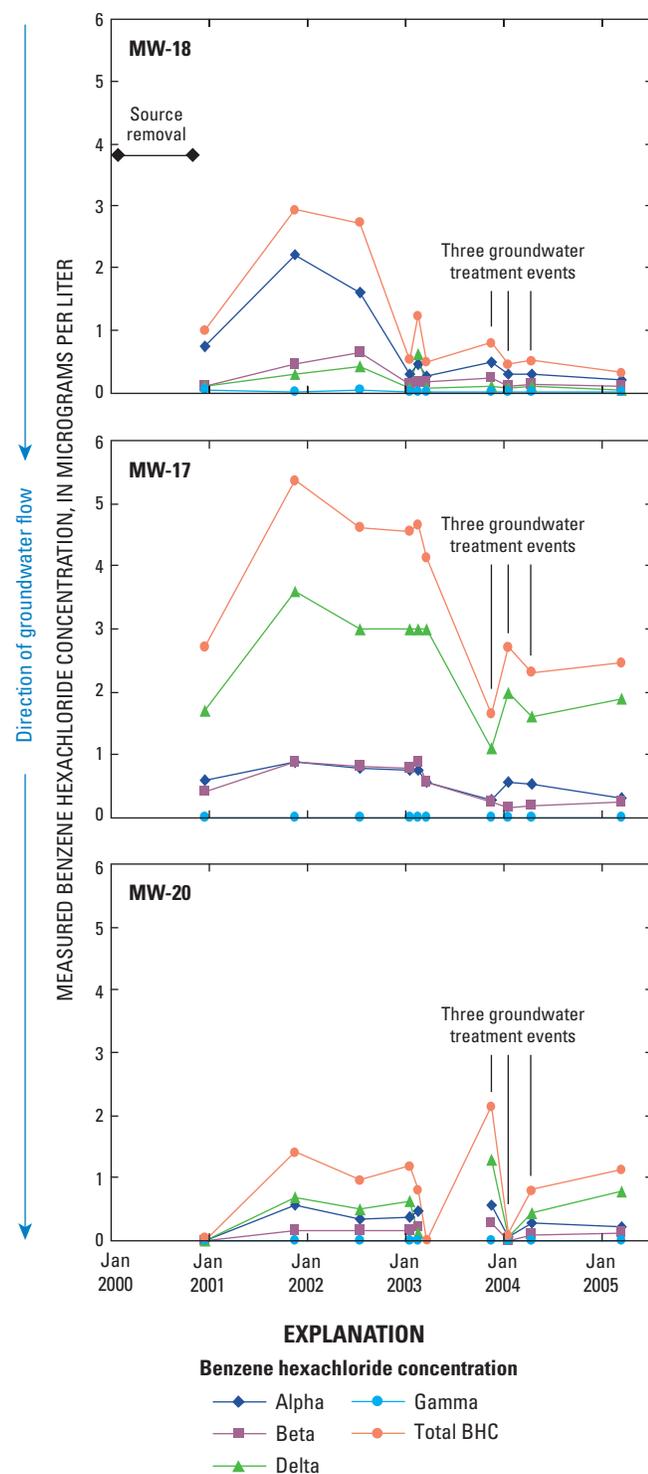
**EXPLANATION**

- Oatland Island Education Center
- Wetland
- Model domain
- Total benzene hexachloride concentration during 2003, in micrograms per liter**
- 0.02
- 0.2
- 1
- Monitoring well



**Figure 2.** Study area and model domain, Oatland Island Education Center, Oatland Island, Georgia.

During a survey of public and private groundwater wells in the area, the following were identified: one potable water well at the site, two public water-supply wells within 2 miles of the site, and at least 12 private drinking water wells within ½ mile



**Figure 3.** Benzene hexachloride (BHC) concentrations in groundwater measured at monitoring wells MW-18, MW-17, and MW-20 for site sampling events, 2000–2005 [MW, monitoring well; sources for data: MWH 2002a, MWH 2002b, MWH 2005].

of the site (S&ME 1999b). The water-supply well at the Oatland site and several private (residential) water-supply wells down-gradient of the site are screened in the Upper Floridan aquifer, several hundred feet bgs. Intervening clay layers (confining units) effectively prevent groundwater and contaminant migration from the surficial to the Upper Floridan aquifer. During site investigations, groundwater samples were collected from onsite and selected off-site water-supply wells screened in the Upper Floridan aquifer, to ensure that these wells were not affected by site contaminants. Samples were analyzed for metals, pesticides, volatiles, and semi-volatile chemicals. None of the chemicals exceeded established detection limits (S&ME 1999b).

## Remediation Activities

In response to the site investigation results, CDC conducted source removal at Area A during January–March and August–October 2000. Source removal included excavation of buried containers and contaminated soil down to the water table. The footprint of the excavated area was about 13,500 square ft (MWH 2001). A total of 5,215 tons of contaminated material were removed from the source area (MWH 2001).

As part of voluntary remediation activities, CDC tested an in-situ groundwater remediation technique recommended by consultants. Two pilot-scale in-situ treatment studies were conducted at the site during 2003. Full-scale treatment began with an initial injection of a modified Fenton’s Reagent (hydrogen peroxide and iron catalysts) during December 2003, followed by additional injections during January and March 2004 (MWH 2005). The intent of the in-situ treatment was to induce chemical degradation of the organochlorine pesticide contaminants in groundwater at the site. Details of the groundwater treatment events are documented in the consultant’s reports (MWH 2003, MWH 2005).

During annual groundwater monitoring in 2005 (after source removal and in-situ treatment), the maximum concentration of total BHC detected in the surficial aquifer was 2.47  $\mu\text{g/L}$  at monitoring well MW-17 (Figures 2 and 3). Individual isomer concentrations ranged from less than 0.27  $\mu\text{g/L}$  for gamma-BHC to 1.9  $\mu\text{g/L}$  for delta-BHC (MWH 2005).

## Environmental Regulatory and Public Health Framework

BHC concentrations in groundwater at the site exceed applicable Georgia Department of Natural Resources, Environmental Protection Division (Georgia EPD) regulatory levels that range from 0.006  $\mu\text{g/L}$  to 0.66  $\mu\text{g/L}$  for the individual BHC isomers (Georgia EPD 1993, MWH 2002a, MWH 2005). Several of the action levels for individual BHC isomers are below the detection limit of 0.044  $\mu\text{g/L}$  for BHC in groundwater. In such cases, the detection limit should be used for regulatory compliance (Georgia EPD 1993). Relevant regulatory action levels and health guidance for BHC in groundwater

at the site are summarized in Table 1, and a discussion of the regulatory and public health framework is presented in the following paragraphs.

The Georgia EPD Hazardous Site Response Act (HSRA) rules apply to releases of contaminants in Georgia. Cleanup criteria under the HSRA rules depend upon current and future land use and reasonable exposure scenarios. For groundwater contaminants, the most stringent cleanup requirements are the Type I risk reduction standards (RRSs) listed in tabular form within Appendix III of Georgia Rule 391-3-19 (Georgia EPD 1993). Georgia EPD Type II RRSs are less stringent for some sites because they can be based on site-specific risk assessments and are not necessarily bound by default exposure factors (Georgia EPD 1993). For BHC contaminants in groundwater at Oatland Island, the Type II RRSs were calculated by Montgomery Watson Harza Americas Inc. (MWH) consultants using a target excess cancer risk of 1 in 100,000 ( $10^{-5}$ ) and standard default assumptions, as specified in Georgia Rule 391-3-19 (MWH 2005). Among the BHC isomers, alpha-BHC has the lowest Type I RRS and the lowest Type II RRS, at 0.006  $\mu\text{g/L}$  and 0.028  $\mu\text{g/L}$ , respectively (Table 1). Both the Type I and Type II RRSs for alpha-BHC are below the analytical detection limit; therefore, in accordance with Georgia EPD rules, the detection limit should be used for determining compliance (Georgia EPD 1993).

Detection limits for a chemical are dependent on the complexity of the sample matrix (the liquid or solid material that contains the chemical) and the performance characteristics (precision and accuracy) of the analytical method used. Method detection limits are defined by the analytical labora-

tory when a particular sample is analyzed. Typical detection limits cited for gamma-BHC in groundwater range from 0.0128  $\mu\text{g/L}$  to 0.044  $\mu\text{g/L}$  (USEPA 1993, Wisconsin Department of Natural Resources [DNR] 1998, Environmental Micro Analysis Inc. [EMA] 2006). It is reasonable to assume that typical detection limits for the other BHC isomers are similar. For compliance purposes, the U.S. Environmental Protection Agency (USEPA) Office of Ground Water and Drinking Water proposed 0.044  $\mu\text{g/L}$  as an acceptable detection limit for gamma-BHC in water samples (USEPA 1993; D. Munch, USEPA Office of Groundwater and Drinking Water/Office of Research and Development, personal communication, November 2006).

As noted previously, the shallow groundwater at the site is not used for drinking water; however, for reference, the USEPA maximum contaminant level (MCL) and the ATSDR comparison values for ingestion of BHC in water are provided in Table 1. USEPA MCLs are legally enforceable contaminant levels that apply to public water-supply systems. MCLs represent contaminant concentrations in drinking water that USEPA deems attainable (using best available treatment technology and cost considerations) and are protective of public health over a lifetime of exposure (70 years). Only gamma-BHC, with an MCL of 0.2  $\mu\text{g/L}$ , is currently regulated under USEPA's national primary drinking-water standards (Table 1).

ATSDR comparison values are concentrations of a chemical in water, soil, or air to which humans may be exposed without experiencing adverse health effects. Comparison values are used for screening purposes only and are, by design, highly conservative to allow early identification of

**Table 1.** Reference values for regulatory action and health guidance for benzene hexachloride in groundwater, Oatland Island, Georgia.

[BHC, benzene hexachloride; Georgia EPD, Georgia Environmental Protection Division; RRS, risk reduction standards;  $\mu\text{g/L}$ , microgram per liter; USEPA, United States Environmental Protection Agency; MCL, maximum contaminant level; ATSDR, Agency for Toxic Substances and Disease Registry; —, a reference value is not available]

BHC isomer	Georgia EPD		USEPA drinking water MCL <sup>3</sup> ( $\mu\text{g/L}$ )	ATSDR comparison value <sup>4</sup> ( $\mu\text{g/L}$ )	Typical detection limit <sup>5</sup> ( $\mu\text{g/L}$ )
	Type I RRS <sup>1</sup> ( $\mu\text{g/L}$ )	Type II RRS <sup>2</sup> ( $\mu\text{g/L}$ )			
Alpha-BHC	0.006	0.028	—	0.006	0.044
Beta-BHC	0.02	0.097	—	0.02	0.044
Delta-BHC	—	0.66	—	0.1	0.044
Gamma-BHC	0.2	0.66	0.2	0.02	0.044

<sup>1</sup> Defined in Georgia Rule 391-3-19.07, Appendix II, Table 1 (Georgia EPD 1993).

<sup>2</sup> Calculated by a consultant using site-specific reasonable maximum exposure scenarios as defined in Georgia Rule 391-3-19.07(7) (MWH 2005, Georgia EPD 1993).

<sup>3</sup> USEPA national primary drinking water maximum contaminant levels are available online at <http://www.epa.gov/safewater/mcl.html>.

<sup>4</sup> ATSDR comparison values are described in the body of this report and in the online publication entitled "ATSDR Public Health Assessment Guidance Manual," available at <http://www.atsdr.cdc.gov/2p-publications.html>.

<sup>5</sup> Detection limits for a chemical vary, depending on matrix complexity and analytical method performance. The USEPA Office of Groundwater and Drinking Water proposed 0.044  $\mu\text{g/L}$  as an acceptable detection limit for gamma-BHC in water samples (USEPA 1993; D. Munch, USEPA Office of Groundwater and Drinking Water/Office of Research and Development, personal communication, November 2006). Similar detection limits were not available for the alpha-, beta-, and delta-BHC isomers, therefore, the 0.044  $\mu\text{g/L}$  detection limit was inferred for these isomers due to their similarity to gamma-BHC.

potential health problems. It is important to note that environmental chemical concentrations alone do not determine the probability of adverse health effects in an exposed population. Rather, the actual probability of adverse effects is dependent on many other factors such as site-specific conditions, the route, magnitude, and duration of actual exposure, and individual lifestyle and genetic factors. Alpha-BHC has the lowest ATSDR comparison value (0.006  $\mu\text{g/L}$ ) among the four BHC isomers (Table 1).

## Contaminant Properties and Health Effects

BHC is more accurately termed hexachlorocyclohexane (HCH), in accordance with International Union of Pure and Applied Chemistry nomenclature recommendations. BHC belongs to the organochlorine class of pesticides, with a chemical structure consisting of a benzene ring and six chlorine atoms. BHC exists in eight chemical forms called isomers that differ only in the spatial arrangement of the chlorine atoms around the benzene ring. Only four of the BHC isomers, the alpha, beta, delta, and gamma forms, are commercially significant.

Technical-grade BHC is a synthetic mixture of several isomers, typically 60–70% alpha-BHC, 5–12% beta-BHC, 10–15% gamma-BHC, 6–10% delta-BHC, and 3–4% epsilon-BHC (ATSDR 2005b). Almost all of the insecticidal properties reside in the gamma isomer; therefore, it is often distilled from the technical-grade mixture into a 99% pure form, which is commercially known as lindane.

Organochlorine compounds were developed after World War II for pest eradication and control. Gamma-BHC (lindane) and technical-grade BHC were produced and used as a pesticide on fruit, vegetable, and forest crops, and animals and animal premises in the past (ATSDR 2005b). BHC is no longer produced in the United States, and most registered agricultural uses have been banned or voluntarily cancelled (USEPA 2006a). It is still used as an active ingredient in some pharmaceutical products formulated for lice and scabies (mites) treatment in humans (ATSDR 2005b, USEPA 2006a). Because of widespread past and current use, BHC has been detected in air, soil, sediment, surface water, groundwater, ice, fish, wildlife, and humans (USEPA 2006b).

As with most organochlorine pesticides, acute exposure to BHC affects the central nervous system in humans, causing dizziness, headaches, convulsions, and seizures. Exposure to BHC also can affect the liver and kidneys and can cause blood disorders. Animal studies indicate that all BHC isomers may cause cancer in humans (ATSDR 2005b). However, USEPA classifies only alpha-BHC and technical-grade BHC as probable carcinogens and beta-BHC as a possible carcinogen. USEPA determined that the evidence is not sufficient to assess or classify the carcinogenic potential of gamma-BHC and delta-BHC.

Two important contaminant properties to consider for this study are the aqueous (water) solubility and density of BHC. BHC is not very soluble in water. Citations for aqueous

solubility of the BHC isomers vary, but typical values are in the range of 5–17 milligrams per liter (mg/L) (ATSDR 2005b). The density of the BHC isomers is 1.87–1.89 grams per cubic centimeter ( $\text{g/cm}^3$ ) at 20 degrees Celsius (ATSDR 2005b), making them denser than water (water density is 1.0  $\text{g/cm}^3$ ).

Because BHC isomers are not very soluble in water and are denser than water, they are considered dense non-aqueous phase liquids (DNAPLs). When a DNAPL is released into groundwater, it can migrate downward (vertically) as a separate phase liquid, leaving residual DNAPL in soil pore spaces below the groundwater surface. Residual DNAPL can act as an ongoing source of groundwater contamination as it partitions into the groundwater slowly, over a long period of time. Residual DNAPL also can migrate downward (vertically) in an aquifer until it encounters a confining unit such as a clay layer, where it accumulates in pools (Pankow and Cherry 1996, Schwillie 1988).

## Methods

ATSDR used an analytical model to estimate organochlorine pesticide migration in shallow groundwater at Otland Island. Both deterministic (“single-point”) and probabilistic (Monte Carlo) analyses were conducted to provide a robust estimate of pesticide migration that includes the variability and uncertainties inherent in groundwater flow and contaminant transport.

The analytical contaminant transport analysis system (ACTS) version 7.8 software used in the analysis is publicly available.<sup>1</sup> ACTS contains more than 100 models and associated analytical solutions for assessing fate and transport of contaminants within four environmental transport pathways—air, soil, surface water, and groundwater. Monte Carlo simulation is integrated into the ACTS software so there is no need to export simulation results into a separate software application. Several publications are available that detail the capabilities of ACTS and present case studies for its application in the context of environmental health (Maslia and Aral 2004, Maslia et al. 1997, Rodenbeck and Maslia 1998).

## Modeling Approach

ACTS was applied to the surficial aquifer using a two-dimensional, saturated aquifer contaminant fate and transport model with constant dispersion coefficients. Groundwater monitoring for Area A, conducted annually since 2000, indicates steady, uniform flow in the surficial aquifer (MWH 2003, MWH 2005). Given these conditions, ATSDR’s two-dimensional analysis considers contaminant transport due to advection (movement in the direction of the bulk

<sup>1</sup> ACTS software can be downloaded from the Georgia Institute of Technology, School of Civil and Environmental Engineering, Multimedia Environmental Simulations Laboratory Web site: <http://mesl.ce.gatech.edu/>

groundwater flow) and horizontal dispersion (movement in a horizontal plane that is parallel to and perpendicular to the groundwater flow direction).

The model domain, encompassing the area of interest at the site, was selected as a grid with a longitudinal length of 850 ft (x-axis) and a transverse length of 500 ft (y-axis) (Figure 2). The x-axis of the model grid is aligned parallel to groundwater flow, and the y-axis is centered on the former pesticide source area.

## Assumptions

For modeling purposes, a number of assumptions were made:

- Groundwater flow is steady and uniform. This assumption is supported by field measurements and analysis of the potentiometric surface (MWH 2003, MWH 2005).
- Tidal effects within the study area would primarily increase contaminant dispersion. This is accounted for in the probabilistic model by varying the longitudinal and transverse dispersion coefficients. Field data collected over a full tidal cycle indicated water level fluctuations of only 0.04 ft in monitoring wells within Area A (MWH 2001).
- The surficial aquifer is characterized as infinite with a constant dispersion coefficient. The infinite aquifer assumption means that aquifer boundaries are far enough away that they do not affect groundwater flow and contaminant transport in the area of interest.
- The time required for leakage and vertical transport of contaminants from the buried containers to the groundwater is negligible. This is consistent with the shallow depth to groundwater at the site (the surficial aquifer occurs at 5–9 ft bgs) and the types of containers excavated from the site (metal containers, cardboard boxes, plastic bags) (MWH 2001, S&ME 1999b).
- The time required for vertical transport of contaminants through the thickness of the surficial aquifer is negligible; the contaminants are well-mixed throughout the 14-foot depth.
- Vertical transport (sinking) of the dissolved phase contaminant plume due to contaminant density effects is negligible. This is a valid assumption given the low aqueous solubility of the contaminants at the site (Schwille 1988).
- Biological and chemical degradation of the contaminants are negligible. This is a conservative assumption. Various studies indicate the half-life for gamma-BHC in water (groundwater and sterilized natural water) may range from 32 to more than 300 days (ATSDR 2005b, Mackay et al. 1997).

## Reference Contaminant Selection

Total BHC concentrations in groundwater were used to calibrate the model and conduct fate and transport simulations of the contaminant plume at Area A. Model results for total BHC are considered representative of all four BHC isomers detected at the site. For modeling purposes, 0.044 µg/L was used as the detection limit and default compliance goal for total BHC in groundwater (Georgia EPD 1993; D. Munch, USEPA Office of Groundwater and Drinking Water/Office of Research and Development, personal communication, November 2006).

Selecting BHC—specifically total BHC—for model calibration and transport simulations is a key decision that deserves a note of explanation. ATSDR selected BHC as the reference contaminant because the other organochlorine pesticides detected at the site, including aldrin, dieldrin, and DDT-related compounds, are much less mobile than BHC in groundwater and have been detected only in the source area where pesticides and associated materials were buried. Therefore, BHC would migrate the farthest and the fastest. The earliest expected time of BHC arrival at the wetlands boundary would be the most critical in terms of ecological risk and impact.

ATSDR used total BHC instead of individual BHC isomers for the modeling because the observed transport behavior of the individual BHC isomers—relative to one another—is not consistent with expected behavior, based on the physical-chemical properties of each. Specifically, available literature values for partition coefficients<sup>2</sup> and calculated retardation factors suggest that alpha-, beta-, and gamma-BHC have similar mobility in groundwater, while delta-BHC has a much lower mobility. Based on measured contaminant concentrations at the site, however, delta-BHC exhibits the highest mobility while gamma-BHC appears to be the least mobile of the four isomers (Figure 3). Several factors could contribute to the difference between expected and observed contaminant mobility:

- Site characteristics may be different than those studied in literature citations for partition coefficients. Differences in soil and groundwater chemistry, including the presence of dissolved organic matter, can have a substantial effect on contaminant fate and transport (Chiou 2002).
- The available partition coefficients for delta-BHC may be less accurate than for the other isomers. Delta-BHC is not the most active isomer in terms of insecticidal properties (gamma-BHC is the most active), and it is not the most prevalent isomer created during synthesis (alpha-BHC composes 60–70% of the synthetic mixture while delta-BHC composes only 6–10%).

<sup>2</sup> Partition coefficients are an experimental measure of how a chemical distributes between two phase systems, such as octanol and water ( $K_{ow}$ ). Measured partition coefficients for a chemical are used to calculate a soil distribution coefficient ( $K_d$ ) and a retardation factor ( $R$ ) that quantifies how much the chemical may be slowed down during groundwater transport because of its tendency to sorb to soil particles.

As a result, delta-BHC is the least studied of the BHC isomers, with only a limited number of partition coefficients cited in the literature (Table B1).

- Recent USGS research highlights significant uncertainties and inaccuracies in literature citations for partition coefficients of even widely studied compounds, such as DDT (Pontolillo and Eganhouse 2001).

Partition coefficients are used to calculate contaminant retardation factors needed for model input. Therefore, if the model is calibrated using the measured concentrations and documented partition coefficient values of one of the individual isomers—delta-BHC, for example, because it has the highest concentration at the leading edge of the plume—it will not necessarily match the transport behavior of one or more of the other isomers. To address this, ATSDR calibrated the model to total BHC concentration (which is predominantly delta-BHC at the leading edge of the plume [Figure 3, Figure A4]) by selecting partition coefficient and retardation coefficient values for the model based on the published ranges of the more studied isomers, alpha-BHC and gamma-BHC. Additionally, for probabilistic transport simulations, ATSDR varied the contaminant retardation coefficient over a range of published values for the BHC isomers.

## Model Input Parameters and Source Definition for Deterministic Simulations

Media, contaminant, and model parameter values used to simulate the fate and transport of total BHC in groundwater for deterministic analyses are summarized in Table 2. ATSDR obtained these input parameter values from a variety of sources, including existing site field data and modeling analyses, available USGS information for the area, and relevant literature articles and other published references on groundwater flow and contaminant transport. Detailed information and notes regarding many of the media and contaminant parameters are included in Appendices B and C.

Contaminant source characteristics were defined in the model based upon existing contaminant and site information. The source area concentration of total BHC is modeled as a Gaussian distribution in the lateral direction (along the y-axis of the model grid). The maximum width of the source perpendicular to groundwater flow is assumed to be 150 ft, based on the horizontal footprint of the source area excavation (MWH 2001). Initial source concentration values of 10,000 µg/L, 7,000 µg/L, 1,000 µg/L, 500 µg/L, 400 µg/L, 200 µg/L, and 100 µg/L total BHC in groundwater were evaluated during model calibration. Initial concentration values of 400 µg/L and 200 µg/L total BHC offered the best calibration results. An initial concentration of 400 µg/L (0.4 mg/L) total BHC in groundwater at the source was assumed for all model

simulations presented in this report. This concentration is approximately 5% of the solubility limit of BHC in groundwater<sup>3</sup> and two orders of magnitude higher than the maximum concentration of total BHC detected in groundwater to date.<sup>4</sup>

Two scenarios were used within the model to explore the change in total BHC concentration at the source as a result of source remediation during 2000. In the first scenario, it was assumed that the remediation, which included excavation of pesticide containers and contaminated soil down to groundwater, effectively eliminated the contaminant source. The eliminated source scenario is characterized in the model as a stepped reduction in source concentration over a 1-year period, with an initial source concentration of 400 µg/L at a simulation time of 30 years (2000), an interim source concentration of 200 µg/L at a simulation time of 30.5 years, and a final source concentration of 0.0 µg/L at a simulation time of 31 years (2001) (Table 3).

In the second scenario, the source concentration of total BHC in groundwater was modeled as constant. The constant source scenario is characterized in the model as a sustained source concentration of 400 µg/L during the entire simulation period (Table 3). This model scenario represents site conditions that may have resulted if no source remediation occurred. The constant source scenario also could represent post-remediation site conditions where residual DNAPL is present in the source area below the water table. Residual DNAPL could act as an ongoing source to groundwater until the residual DNAPL is depleted. This is a purely hypothetical scenario; site-specific data are not available to confirm or quantify DNAPL in the source area.

## Model Calibration Using Plume-Matching

The analytical model was calibrated by manually adjusting input parameters to achieve simulation results that matched contaminant concentrations measured in site monitoring wells. Simulation results were compared to field measurements from three monitoring wells located along the centerline of the contaminant plume (MW-17, MW-20, MW-23) for longitudinal contaminant transport (Table 4). To calibrate lateral contaminant transport, the maximum contaminant plume width during 2003 was used (Table 4). Field data for determining plume width was obtained from contaminant contours and analytical results documented in consultant's reports (Figure A4, MWH 2003, MWH 2005).

<sup>3</sup> Literature values for aqueous solubility of the BHC isomers vary; see Appendix B.

<sup>4</sup> The maximum concentration of total BHC detected in groundwater was 5.38 µg/L at MW-17 during November 2001 (MWH 2002a). Detections of BHC in groundwater at the site were predominantly below 1 µg/L (MWH 2002a, MWH 2002b, MWH 2005).

**Table 2.** Calibrated parameter values used to simulate the fate and transport of total benzene hexachloride in groundwater for deterministic analysis using ACTS software, Oatland Island, Georgia.

[BHC, benzene hexachloride; ACTS, analytical contaminant transport analysis system software; ft, feet; d, day; mi, mile; lb, pound; ft<sup>3</sup>, cubic feet; yr, year; ft<sup>2</sup>, square feet; in, inch; µg, microgram; mg, milligram; L, liter]

Parameter	Value	Data source <sup>1</sup> or calculation method
Hydraulic conductivity ( $K$ )	9 ft/d	Selected from site-specific (small-scale and large-scale) data and literature values (see Table B1).
Hydraulic gradient ( $i$ )	35.38 ft/mi	Geometric mean of site-specific hydraulic gradients; (S&ME 1999b, MWH 2003, MWH 2005)
Porosity of soil ( $\theta$ )	0.4	Selected from site-specific data and literature values (see Table B1).
Bulk density of soil ( $\rho_b$ )	99.9 lb/ft <sup>3</sup>	Typical for fine sand (Morris and Johnson 1967)
Specific discharge ( $V_d$ )	22 ft/yr	$V_d = Ki$
Groundwater velocity ( $V$ )	55 ft/yr (0.15 ft/d)	$V = Ki / \theta$
Longitudinal dispersivity ( $\alpha_L$ )	27.5 ft	Selected as 5.5% of estimated aquifer length of 500 ft; (Fetter 1993, Gelhar et al. 1992)
Ratio of dispersivities ( $\alpha_L/\alpha_T$ )	100	(Fetter 1993, Gelhar et al. 1992)
Longitudinal dispersion coefficient ( $D_x$ )	1,500 ft <sup>2</sup> /yr	$D_x = \alpha_L V$
Lateral dispersion coefficient ( $D_y$ )	15.0 ft <sup>2</sup> /yr	$D_y = D_x (\alpha_T / \alpha_L)$
Recharge to surficial aquifer ( $q$ )	6 in/yr	Calculated from precipitation, surface runoff, and evapotranspiration data for the area (see Table B1).
Retardation coefficient for BHC ( $R$ )	14.5	Selected value calculated from literature references for BHC partition coefficients (see Table B1).
Source concentration ( $C$ )		Initial concentration in groundwater assumed to be at 5% of the solubility limit for BHC (Table B1). Source removal during 2000 simulated as a stepped reduction.
At time = 0 yrs (1970)	400 µg/L	
At time = 30.5 yrs (June 2000)	200 µg/L	
At time = 31 yrs (2001)	0 µg/L	
Standard deviation ( $\sigma_w$ ) of contaminant source width, assuming a Gaussian distribution	30 ft	For $\sigma_w = 30$ ft, $\pm 2 \sigma_w$ spans 120 ft and encompasses 95% of the source width. The actual width of the source area excavation (perpendicular to groundwater flow) is approximately 150 ft (MWH 2001).
Aquifer thickness	14 ft	(MWH 2003, MWH 2005)
X-coordinate length	850 ft	ACTS computational grid geometry; x-axis aligned with direction of groundwater flow.
Discretization along x-direction	25 ft	ACTS computational grid geometry.
Y-coordinate length	500 ft	ACTS computational grid geometry.
Discretization along y-direction	10 ft	ACTS computational grid geometry.
Location of contaminant source ( $x, y$ )	0.0 ft, 250 ft	ACTS computational grid location.
Location of MW-17 ( $x, y$ )	200 ft, 250 ft	ACTS computational grid location. Distance from source to MW-17 from Figure 4.7 in MWH 2005.
Location of MW-20 ( $x, y$ )	300 ft, 250 ft	ACTS computational grid location. Distance from source to MW-20 from Figure 4.7 in MWH 2005.
Location of MW-23 ( $x, y$ )	500 ft, 250 ft	ACTS computational grid location. Distance from source to MW-23 from Figure 4.7 in MWH 2005.
Location of wetlands ( $x, y$ )	800 ft, 250 ft	ACTS computational grid location. Distance from source to wetlands from Figure 4.7 in MWH 2005.
Duration of simulation	300 yr	ACTS simulation period.
Temporal discretization	0.5 yr	Selected time step for ACTS simulations.

<sup>1</sup>Notes on parameter values are included in Appendix B.

**Table 3.** Contaminant source characterization scenarios.

[BHC, benzene hexachloride; yrs, years; %, percent]

Model simulation time (t)	Simulated source concentration of total BHC in groundwater (micrograms per liter, µg/L)	
	Eliminated source scenario	Constant source scenario
t = 0 yrs (1970)	<sup>1</sup> 400	<sup>1</sup> 400
t = 30 yrs (2000)	400	400
t = 30.5 yrs (mid-2000)	200	400
t = 31 yrs (2001)	0	400
t = 70 yrs (2040)	0	400
t = 300 yrs (2270)	0	400

<sup>1</sup>Initial source concentration estimated as 5% of the aqueous solubility for BHC. Literature citations of aqueous solubility for BHC range from 140 to 17,000 µg/L; the mean of 70 citations was calculated as 7,000 µg/L (Clayton and Clayton 1981, Kurihara et al. 1973, Hazardous Substances Data Bank (HSDB) 2003, Hollifield 1979, U.S. Department of Agriculture 2004, Mackay et al 1997).

## Model Input Parameters for Probabilistic Simulations

A two-stage Monte Carlo simulation was used to conduct probabilistic analyses of contaminant transport. In stage 1, probability density functions (PDFs) were developed for eight input parameters—referred to as variants—using Monte Carlo simulation (Table 5). In stage 2, values from each PDF were randomly selected as input parameters for the fate and transport model to generate contaminant concentrations in groundwater for a specific time and location at the site (e.g., selected monitoring wells or the wetlands boundary).

Each PDF consists of a range of 10,000 random values (or realizations) that characterizes the variability and uncertainty of a given input parameter. Within ACTS, the type (shape) and the descriptive statistics (mean, minimum, maximum, variance) for each PDF were defined on the basis of the available site-specific field data and applicable literature references (Appendix B). The mean values specified for each PDF corresponded to the calibrated single-point parameter values used in the deterministic analysis.

Using the eight PDFs as input, 10,000 Monte Carlo realizations were run within ACTS to generate a range of possible contaminant concentrations in groundwater for a specific time and location of interest. Model simulation results for multiple

**Table 4.** Transport model calibration targets selected from available benzene hexachloride groundwater monitoring results, Oatland Island, Georgia.

[BHC, benzene hexachloride; ft, feet; µg/L, microgram per liter; &lt;, less than; =, equals]

Model calibration target	Distance from source (ft)	2001		2003		2005	
		Total BHC concentration (µg/L)		Total BHC concentration (µg/L)		Total BHC concentration (µg/L)	
		Measured <sup>1</sup>	Simulated	Measured <sup>1</sup>	Simulated	Measured <sup>1</sup>	Simulated
MW-17	200	5.4	10.2	1.6–4.7	11.0	2.5	11.7
MW-20	300	1.4	0.6	0.8–2.2	0.8	1.1	1.0
MW-23	500	<0.055	0.000041	<0.055	0.00011	<0.055	0.00026
Plume width at MW-20	300	Measured plume width at 0.02 µg/L total BHC contour = 170 ft during 2003 Simulated plume width at 0.02 µg/L total BHC contour = 170 ft during 2003					

<sup>1</sup>Site-specific data derived from Figure A4, included in this report, and from consultant reports (MWH 2002a, MWH 2002b, MWH 2005).

**Table 5.** Parameter values used to simulate the fate and transport of total benzene hexachloride in groundwater for a probabilistic (Monte Carlo) analysis, Oatland Island, Georgia.

[BHC, Benzene hexachloride; PDF, probability density function; ft, feet; yr, year; ft<sup>2</sup>, feet squared; in, inches; d, day; —, PDF was not generated]

Parameter or variant	Mean value <sup>1</sup>	Range <sup>1</sup> used to generate PDF	PDF type <sup>2</sup>	Statistics for Monte Carlo simulated PDF <sup>3</sup>	
Specific discharge ( $V_d$ )	22 ft/yr	5–98 ft/yr	Lognormal	Minimum: 5.0 Maximum: 97.6	Mean: 21.9 Standard deviation: 10.7
Longitudinal dispersion coefficient ( $D_x$ )	1,500 ft <sup>2</sup> /yr	300–6,000 ft <sup>2</sup> /yr	Lognormal	Minimum: 304.2 Maximum: 5,935.1	Mean: 1,475.8 Standard deviation: 781.0
Lateral dispersion coefficient ( $D_y$ )	15.0 ft <sup>2</sup> /yr	3–60 ft <sup>2</sup> /yr	Lognormal	Minimum: 3.0 Maximum: 59.5	Mean: 15.0 Standard deviation: 8.0
Porosity of soil ( $\theta$ )	0.4	0.3–0.5	Uniform	Minimum: 0.3 Maximum: 0.5	Mean: 0.4 Standard deviation: 0.058
Recharge to surficial aquifer ( $q$ )	0.5 ft/yr (6 in/yr)	0.1–1.5 ft/yr (1–18 in/yr)	Normal	Minimum: 0.1 Maximum: 1.3	Mean: 0.5 Standard deviation: 0.2
Retardation coefficient for BHC ( $R$ )	14	4–22	Triangle	Minimum: 4.2 Maximum: 21.9	Mean: 14.5 Standard deviation: 3.8
Aquifer thickness	14 ft	13–15 ft	Triangle	Minimum: 13.0 Maximum: 15.0	Mean: 14.0 Standard deviation: 0.41
Standard deviation ( $\sigma_w$ ) of contaminant source width, assuming a Gaussian distribution	30 ft	20–40 ft	Normal	Minimum: 20.0 Maximum: 40.0	Mean: 30.0 Standard deviation: 3.7
Hydraulic conductivity ( $K$ )	9 ft/d	<sup>4</sup> 2–40 ft/d	—	PDF was not generated for this parameter. <sup>4</sup>	
Groundwater velocity ( $V$ )	55 ft/yr (0.15 ft/d)	<sup>4</sup> 12–245 ft/yr	—	PDF was not generated for this parameter. <sup>4</sup>	

<sup>1</sup>Detailed references and notes on parameter values are included in Appendix B.

<sup>2</sup>10,000 Monte Carlo iterations were performed for each variant to generate a probability density function (PDF).

<sup>3</sup>Graphs and summary statistics for each PDF are included in Appendix D.

<sup>4</sup>Parameter range shown for reference only; hydraulic conductivity and groundwater velocity are not varied explicitly in the model simulations. These two parameters are varied implicitly: hydraulic conductivity in deriving Darcy velocity and groundwater velocity by varying Darcy velocity and porosity.

time periods were aggregated for selected locations (key monitoring wells, wetlands boundary) to illustrate the probability of exceeding certain compliance goals, such as a detection limit or regulatory action level, over time. The probabilistic simulations were conducted using the eliminated source scenario (Table 3).

Four site locations were selected for probabilistic fate and transport modeling: three site monitoring wells located along the centerline of the contaminant plume (MW-17, MW-20, and MW-23) and the wetlands boundary (Figure 2). The location of each of these selections in relation to the source area is as follows:

- MW-17, located approximately 200 ft downgradient from the source area, is within the contaminant plume. BHC isomers have been detected at this monitoring well during annual sampling events since 2000.

- MW-20, located approximately 300 ft downgradient from the source area, is within the contaminant plume. BHC isomers have been detected at this monitoring well during annual sampling events since 2000.
- MW-23, located along Barley Drive, approximately 500 ft downgradient from the source area, has not been affected by the contaminant plume. BHC has never been detected at this monitoring well during annual sampling events.
- The wetlands boundary, located approximately 800 ft downgradient from the source area, has not been affected by the BHC contaminant plume. The wetlands extend along Richardson Creek, south and west of the Oatland Island Education Center.

## Results

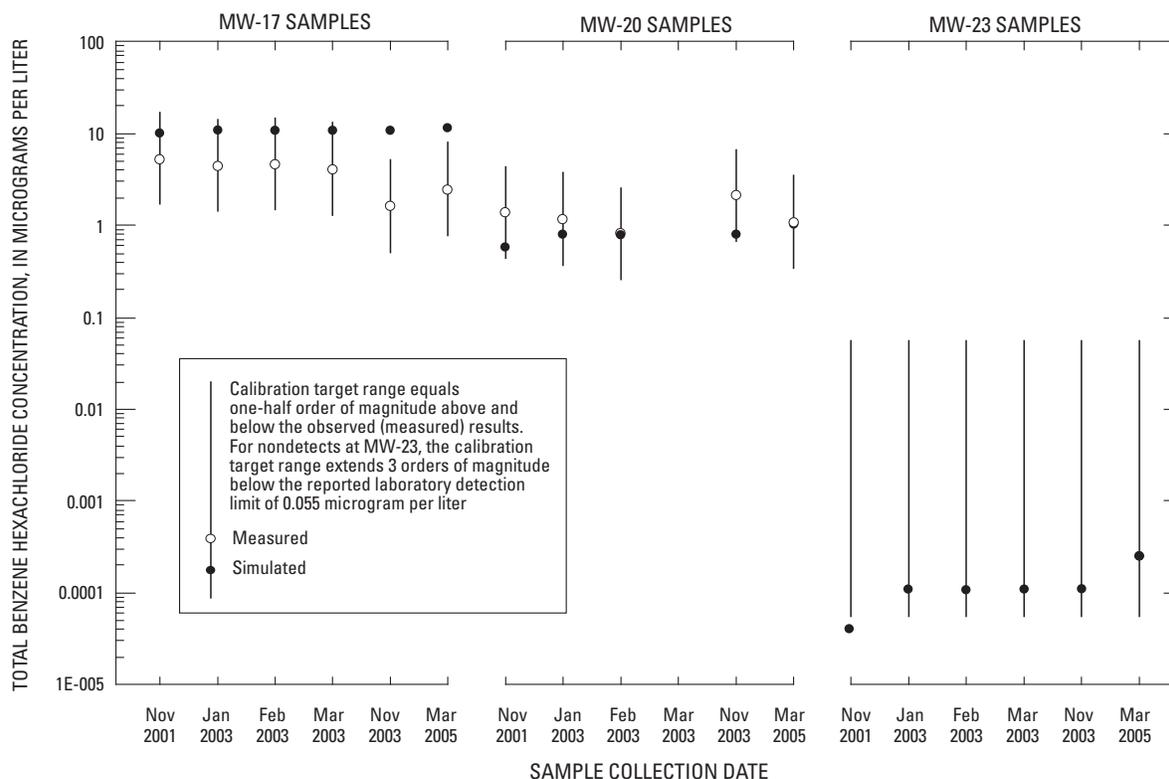
### Model Calibration

Comparisons of measured data and calibrated model simulation results of total BHC in groundwater at wells MW-17, MW-20, and MW-23 for 2001, 2003, and 2005 are presented in Table 4 and Figure 4. Calibration targets for 2003 also are presented graphically in Figure 5 for simulation results expressed as a spatial distribution of total BHC in groundwater.

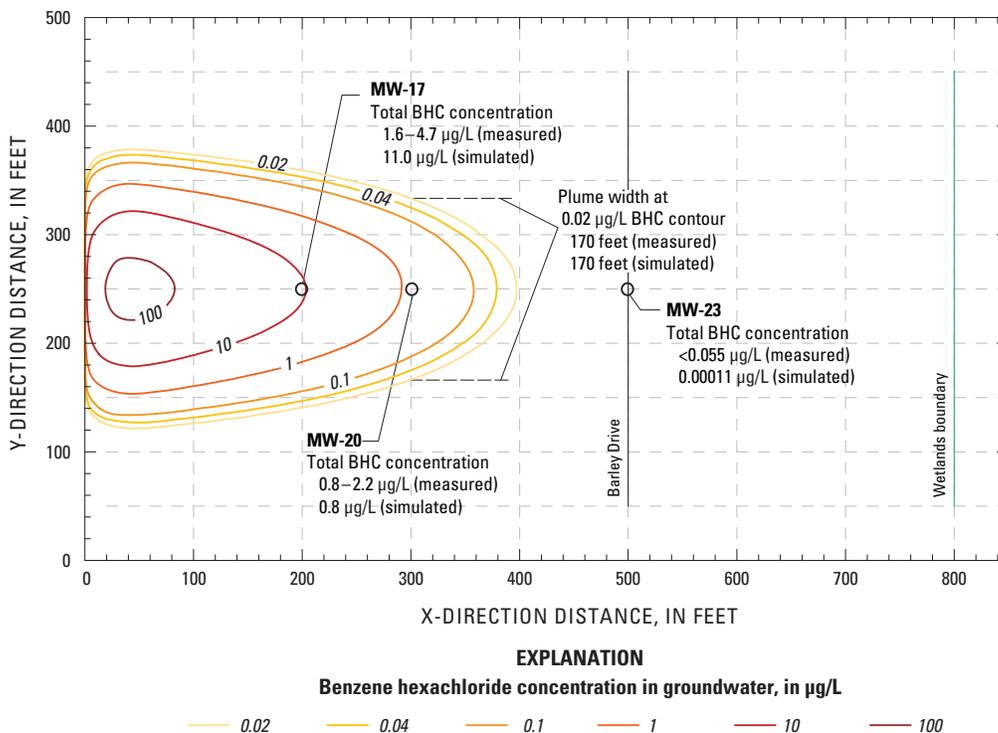
### Deterministic Simulations for an Eliminated Source

Deterministic results using calibrated, single-point values for the input parameters (Table 2) are presented as spatial distributions of total BHC concentrations in groundwater for representative time periods (Figure 6). The simulation times are for 5, 20, 30, 50, 80, and 110 years from the presumed start of contamination (1970). These results are for the eliminated source scenario, where total BHC concentration in groundwater at the source is initially 400  $\mu\text{g/L}$  and decreases to 0.0  $\mu\text{g/L}$  by the end of 2000 (Table 3).

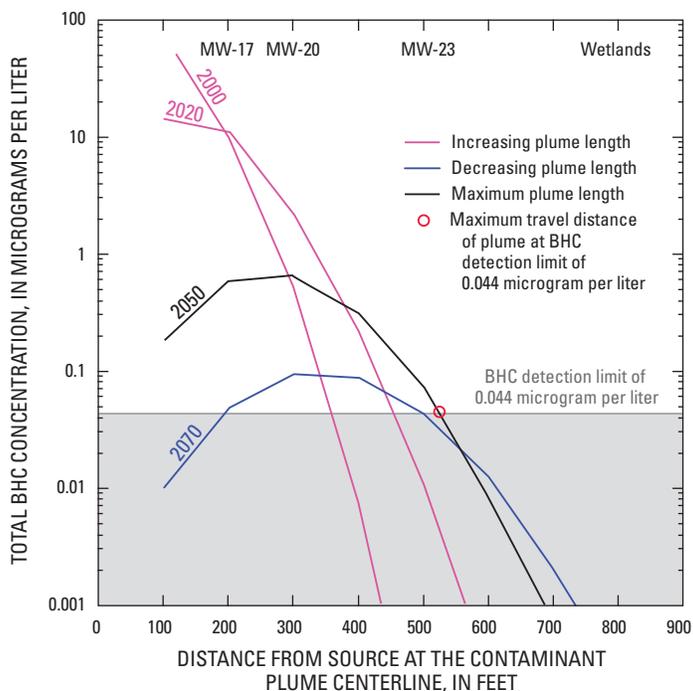
Deterministic results for calibrated input parameters are also presented in Figure 7 as total BHC concentration in groundwater versus distance from the source (along the center-line of the contaminant plume) for various time periods.



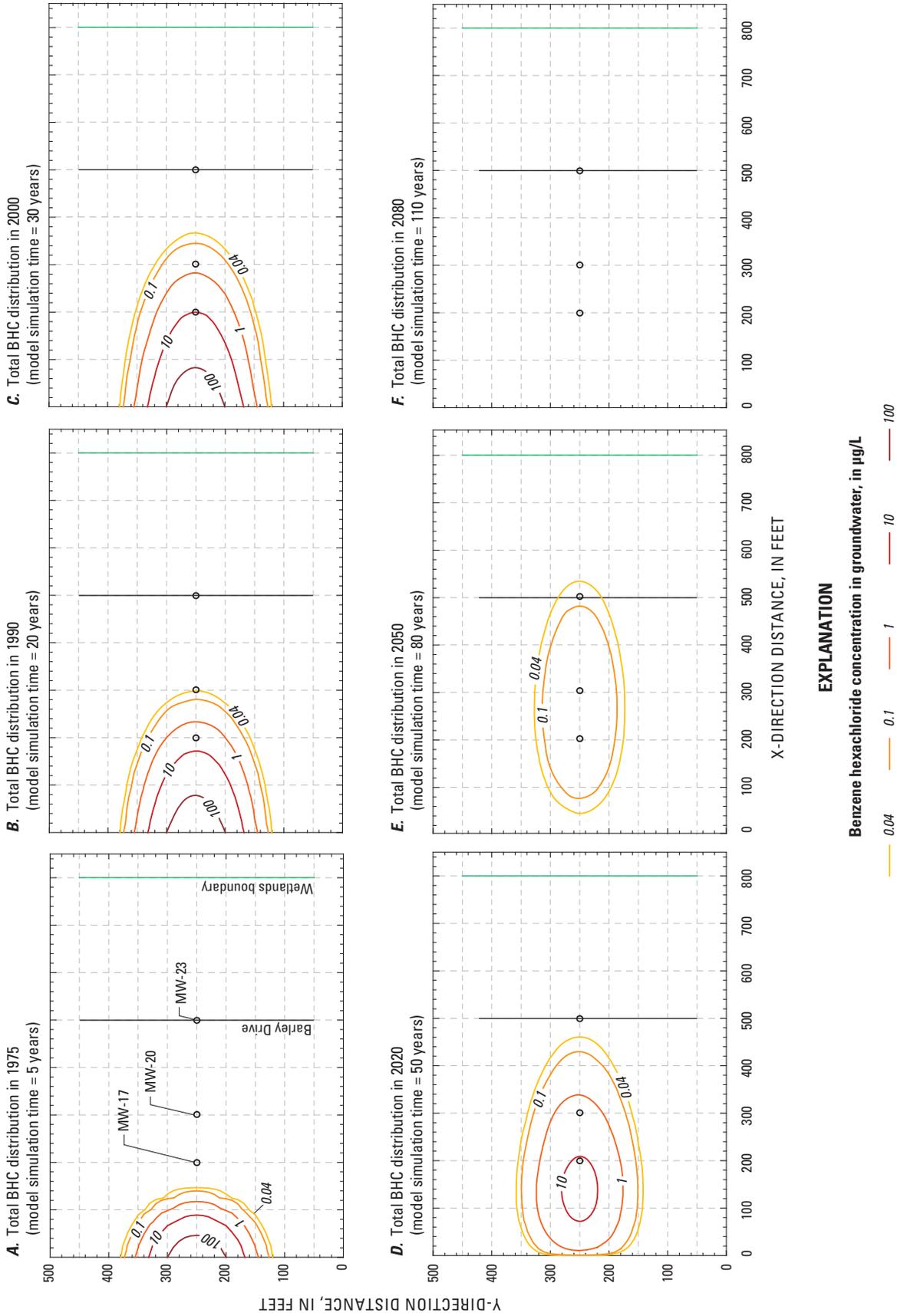
**Figure 4.** Comparison of measured concentrations and deterministic simulation results for total benzene hexachloride concentrations in groundwater for 2001, 2003, and 2005 calibration targets (sources for measured data: MWH 2002a, MWH 2002b, MWH 2005) [MW, monitoring well]



**Figure 5.** Calibration targets superimposed on the spatial distribution of total benzene hexachloride (BHC) concentrations in groundwater for 2003 (model simulation time = 33 years). [MW, monitoring well; µg/L, microgram per liter]



**Figure 7.** Deterministic simulation results of total benzene hexachloride (BHC) concentrations in groundwater versus distance from the source along the contaminant plume centerline during 2000, 2020, 2050, and 2070. [MW, monitoring well]



**Figure 6.** Deterministic simulation results of total benzene hexachloride (BHC) concentrations in groundwater for an eliminated source scenario during (A) 1975, (B) 1990, (C) 2000, (D) 2020, (E) 2050, and (F) 2080. [MW, monitoring well; µg/L, microgram per liter]

## Deterministic Simulations for a Constant Source Probabilistic Type Curves for Each Site Location

Deterministic results for the eliminated source and constant source scenarios are presented for comparison in Figures 8 and 9. The spatial distributions of total BHC concentration in groundwater for representative time periods are shown in Figure 8. Total BHC concentrations versus time for the eliminated and constant source scenarios are presented in Figure 9. The input parameters for each scenario are identical, with the exception of the source characterization outlined in Table 3. In the eliminated source scenario, total BHC concentration in groundwater at the source is initially 400  $\mu\text{g/L}$  and steps down to 0.0  $\mu\text{g/L}$  by the end of 2000. In the constant source scenario, total BHC in groundwater at the source is 400  $\mu\text{g/L}$  for all simulation times.

### Probabilistic Model Simulations

#### Probability Density Functions for Model Variants

The values generated within ACTS for the eight model variants can be expressed as frequency distributions (histograms) or PDFs. Both a histogram and a PDF are shown in Figure 10 for Darcy velocity (specific discharge), one of the eight variants. The histogram (Figure 10A) depicts the frequency, and the PDF (Figure 10B) depicts the relative frequency (an approximation of probability) that each value of Darcy velocity occurs in the set of 10,000 Monte Carlo realizations. The PDFs for each of the eight variants used in probabilistic fate and transport modeling are included in Appendix D. Descriptive statistics for these PDFs are listed in Table 5.

### Probabilistic Results

Results for the probabilistic analysis of total BHC in groundwater at four different site locations (MW-17, MW-20, MW-23, and the wetlands) are depicted in Figure 11 in terms of probability of exceedance versus total BHC concentration in groundwater. These results are for the eliminated source scenario. Selected time periods are shown, beginning in 2005 and progressing to the time period that corresponds to the maximum probability of exceeding the detection limit at each site location. The maximum probability of exceeding the detection limit at each location is clearly shown in a probability of exceedance versus time plot developed specifically for a total BHC concentration of 0.044  $\mu\text{g/L}$  (Figure 12). More information about the type curves in Figure 11 is provided in the following sections describing individual type curves for site locations of interest.

Individual probabilistic type curves were developed for each of the four site locations of interest in the same format as Figure 11, but over a broader range of time periods. The probabilistic type curves for MW-23 and the wetlands are discussed in the following sections and presented graphically in Figures 13 and 14, respectively. Probabilistic type curves for all four locations (MW-17, MW-20, MW-23, and the wetlands) are included in Figures D9–D16. Examples of various probabilistic outputs from ACTS for a single location and time (versus the aggregated results shown on the type curves) also are included in Appendix D.

#### Monitoring Well MW-23

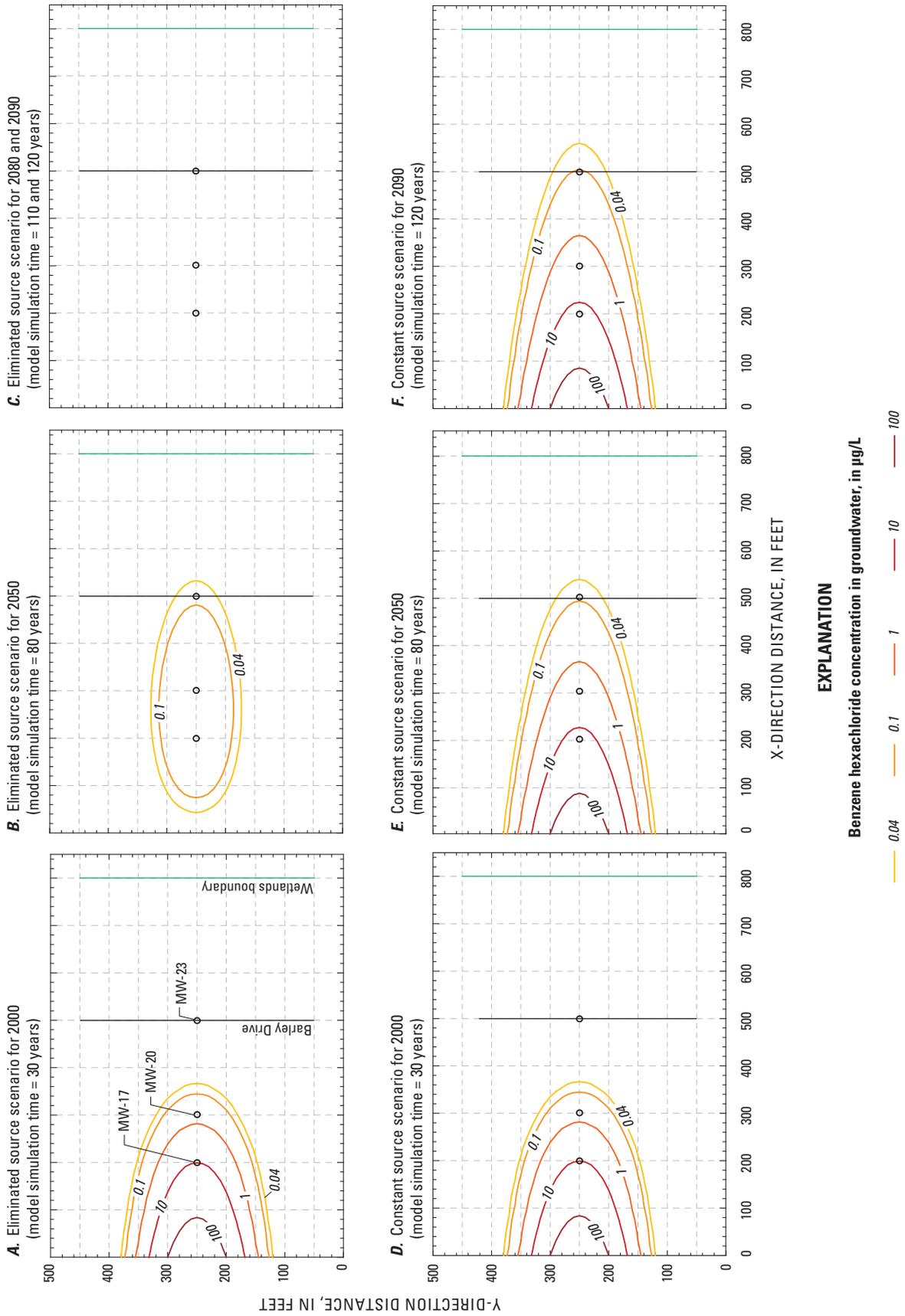
MW-23 is located along Barley Drive approximately 500 ft downgradient from the source area. Probabilistic type curves for MW-23 were developed by aggregating results (concentration of total BHC in groundwater at MW-23) from multiple probabilistic model simulations at different time periods (Figure 13).

Using Figure 13, the probability of exceeding a selected concentration at a given time period is determined by selecting the concentration of interest on the x-axis (for example, 0.044  $\mu\text{g/L}$ ), drawing a vertical line that intersects the selected simulation time curve (for example, 2045), and drawing a horizontal line that intersects the y-axis. At MW-23, the maximum probability of exceeding the detection limit for BHC is 45% during 2045 (Figures 12 and 13).

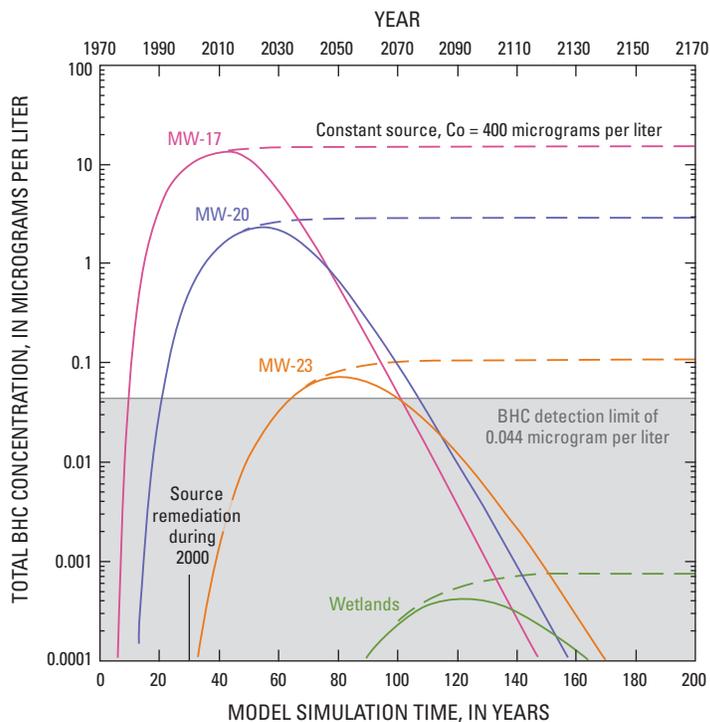
The results for time periods before and after 2045 are presented in separate plots for clarity:

- The probabilistic results included in Figure 13A are presented in terms of probability of exceedance versus total BHC concentration in groundwater for model simulations at 1980, 1990, 2005, 2020, and 2045. The probability of exceeding 0.044  $\mu\text{g/L}$  at MW-23 steadily increases over time from 4% during 1990, to 18% during 2005, to 32% during 2020. The maximum probability of exceeding 0.044  $\mu\text{g/L}$  total BHC at MW-23 is 45% during 2045.
- The probabilistic results included in Figure 13B are presented in terms of probability of exceedance versus total BHC concentration in groundwater for model simulations of 2045, 2070, 2120, 2170, and 2270.

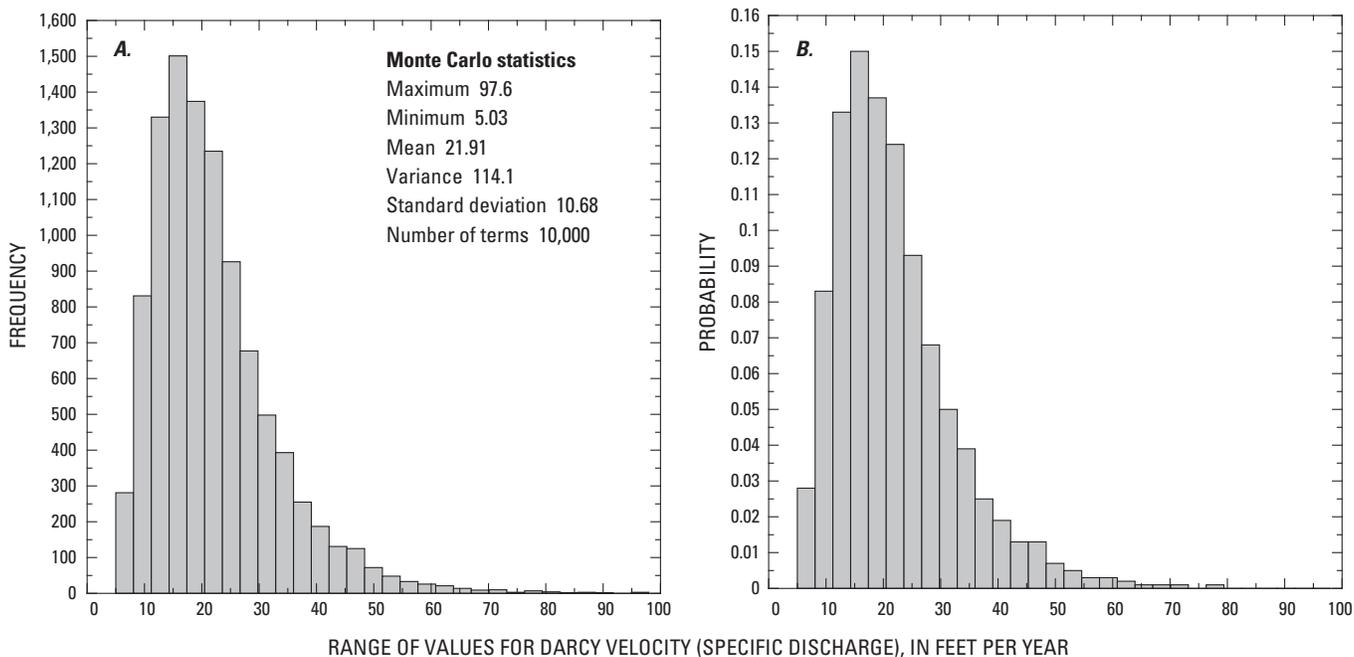
Deterministic model results (from single-point input parameters) are included in Figure 13 for each simulation year depicted, with the exception of 1980 (results for 1980 are below the range of simulated BHC concentrations shown in Figure 13). These results fall in the 40% to 50% probability range because the single-point input parameters used in deterministic simulations are approximately the mean values of the ranges used for probabilistic analysis.



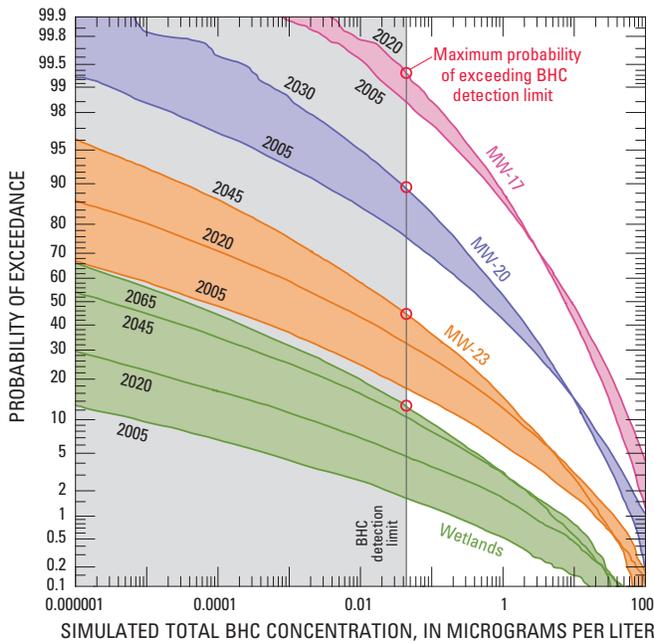
**Figure 8.** Deterministic simulation results of total benzene hexachloride concentrations in groundwater for an eliminated source scenario during (A) 2000, (B) 2050, and (C) 2080 and 2090, and for a constant source scenario during (D) 2000, (E) 2050, and (F) 2090. [MW, monitoring well; µg/L, microgram per liter]



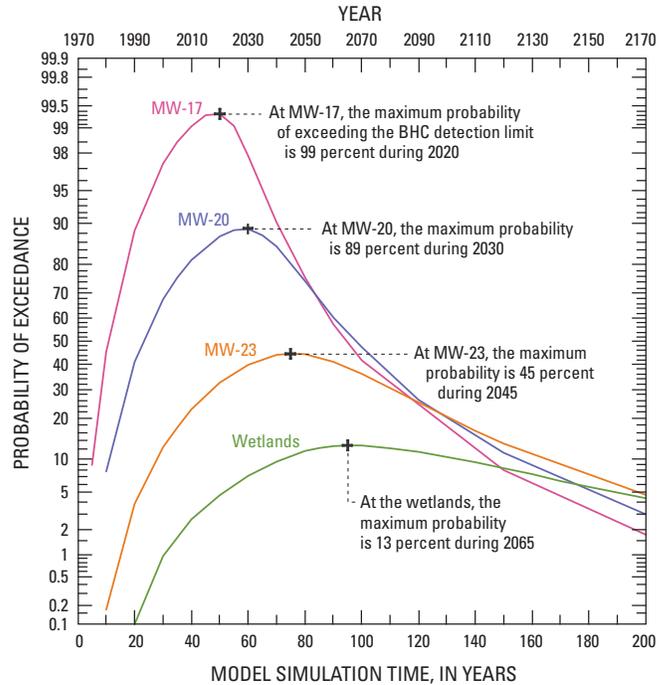
**Figure 9.** Deterministic simulation results of total benzene hexachloride (BHC) concentrations in groundwater over time for an eliminated source scenario (solid lines) and a constant source scenario (dashed lines). [MW, monitoring well]



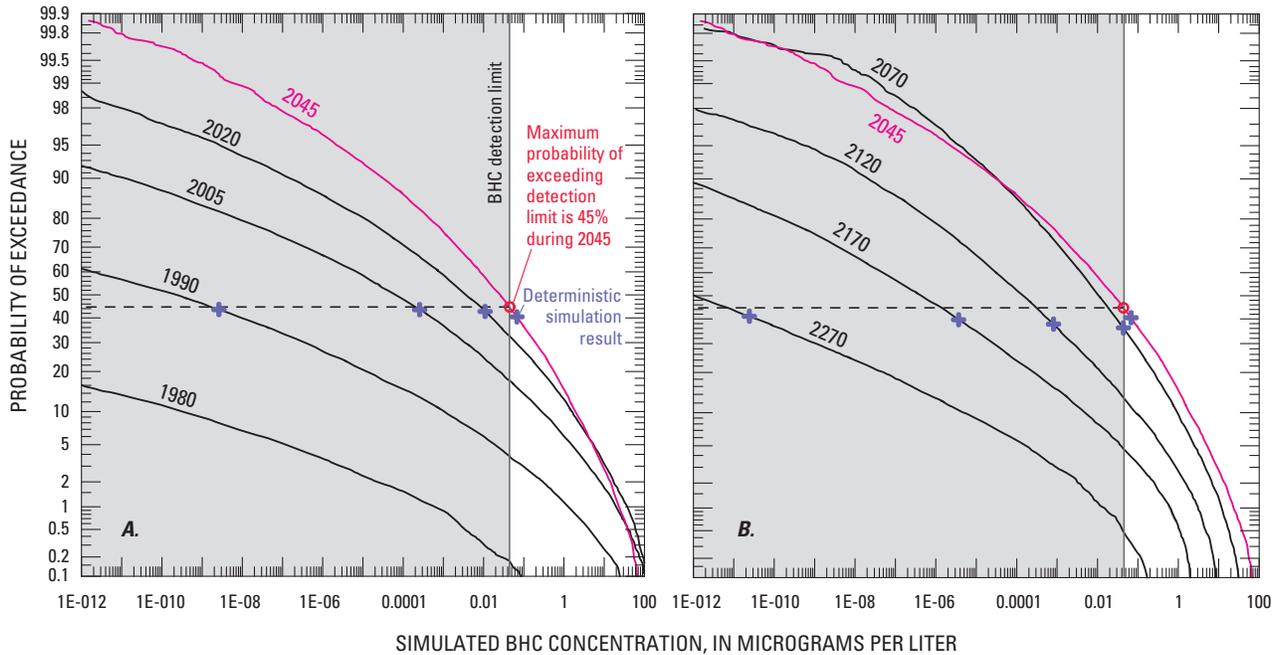
**Figure 10.** Darcy velocity (specific discharge) values generated for probabilistic modeling: (A) frequency distribution (histogram) and (B) probability density.



**Figure 11.** Probabilistic simulation results expressed as probability of exceedance versus simulated total benzene hexachloride (BHC) concentration for four site locations of interest. [MW, monitoring well; BHC detection limit, 0.044 microgram per liter]



**Figure 12.** Probabilistic simulation results expressed as probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter versus simulation time for the eliminated source scenario at four site locations of interest. [MW, monitoring well]



**Figure 13.** Probabilistic model results at MW-23 for time periods (A) before and (B) after 2045, when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [µg/L, microgram per liter; MW, monitoring well]

## Wetlands Boundary

The wetlands boundary is approximately 800 feet down-gradient from the source area. Probabilistic type curves for the wetlands boundary were developed by aggregating results from multiple model simulations at different time periods (Figure 14). At the wetlands, the maximum probability of exceeding the detection limit for BHC is 13% during 2065 (Figures 12 and 14). The results for time periods before and after 2065 are presented in separate plots for clarity:

- The probabilistic results included in Figure 14A are presented in terms of probability of exceedance versus total BHC concentration in groundwater for model simulations of 1980, 1990, 2000, 2020, 2050, and 2065. The probability of exceeding 0.044 µg/L at the wetlands steadily increases over time from 1% during 2000, to 5% during 2020, to 11% during 2050. The maximum probability of exceeding 0.044 µg/L total BHC at the wetlands is 13% during 2065.
- The probabilistic results included in Figure 14B are presented in terms of probability of exceedance versus total BHC concentration in groundwater for model simulations of 2065, 2170, 2220, and 2270.

Deterministic model results (from single-point input parameters) are included in Figure 14 for each simulation year depicted, with the exception of 1980 and 1990 (results for these are below the range of simulated BHC concentrations shown in Figure 14).

## Summary Matrix for Selected Compliance Goals

The summary matrix in Table 6 presents probabilistic results at various compliance goals for MW-20, MW-23, and the wetlands. The time period listed in Table 6 for each site location corresponds to the simulation time where the maximum probability of exceedance occurred.

## Discussion

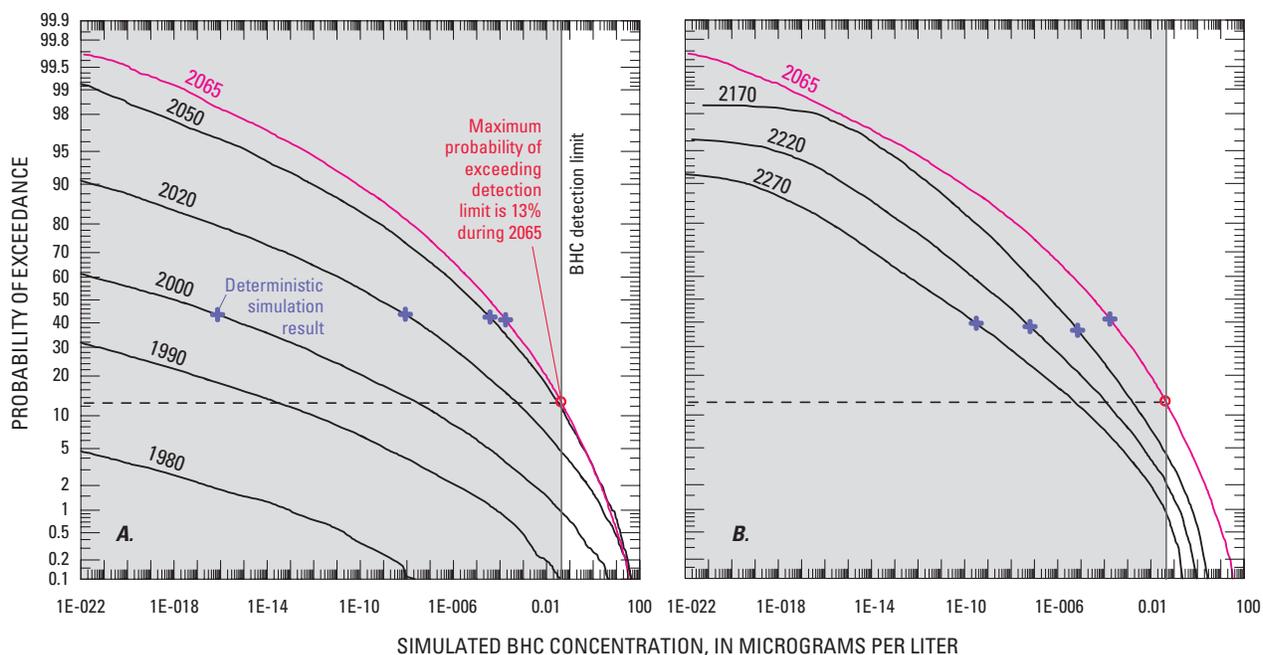
Modeling results obtained by applying ACTS in a deterministic (single-point input) mode are in good agreement with measured groundwater contaminant concentrations at the Oatland Island site. Deterministic model results are slightly higher than measured concentrations at MW-17, but nearly identical at MW-20 and MW-23 (Table 4, Figures 4 and 5). The reason for the greater difference between measured and simulated concentrations at MW-17 could be attributed to greater heterogeneity near MW-17 that is not accurately characterized by a single mean value (deterministic) model input, analytical model limitations, measurement error, or a combination of all of the above. These calibration results are considered quite acceptable for fate and transport simulations using an analytical model.

Both the deterministic and probabilistic simulation results from two-dimensional analytical fate and transport modeling provide insights into the migration of pesticides in groundwater at the Oatland Island site. The deterministic simulations using single-point input parameters indicate the following:

- The BHC contaminant plume develops rapidly during the 30 years that the contaminant source was in place (Figure 6).
- The source removal during 2000 occurred before the contaminant plume reached steady state. Consequently, the leading edge of the contaminant plume continues to migrate downgradient after source removal (Figures 6 and 7).
- In the eliminated source scenario, the maximum travel distance for the leading edge of the contaminant plume (at 0.044 µg/L) is 540 feet after 80 years (2050). The BHC contaminant plume (at 0.044 µg/L) does not advance further downgradient after 80 years (Figures 6 and 7).
- Deterministic simulation results for calibrated, single value input parameters indicate the contaminant plume will not affect the wetlands (Figures 6–9). Simulated total BHC concentrations at the wetlands are below the BHC detection limit (0.044 µg/L) and below all relevant BHC reference values for regulatory and health guidance listed in Table 1.
- Simulated total BHC concentrations in groundwater are above the detection limit (0.044 µg/L) at MW-23 during 2030–2070 (Figure 9).
- In the constant source scenario, the maximum travel distance for the leading edge of the contaminant plume (at 0.044 µg/L) is 560 feet after 120 years (2090). The BHC contaminant plume does not advance further downgradient after 120 years (Figure 8).

Probabilistic simulations using 10,000 realizations for each of eight different input parameters indicate the probability of exceeding the detection limit (0.044 µg/L) of total BHC in groundwater at the wetlands boundary increases from 1% during 2000 to a maximum of 13% during 2065 (Figures 12 and 14). Thus, there is an 87% confidence level that the downgradient wetlands will not be affected by the pesticide contaminant plume.

MW-23 is located along the centerline of the plume. It is halfway between the former source area and the wetlands, it is screened across the full depth of the surficial aquifer, and it has not been affected by the contaminants (no detections of BHC in eight groundwater monitoring events during 2000–2005). Because of its location, MW-23 is particularly well-suited and sensitive as a compliance tool for decision-making. Model simulations predict a 12% to 45% probability that total BHC will be above the detection limit during 2000–2045 at



**Figure 14.** Probabilistic model results at the wetlands for time periods (A) before and (B) after 2065, when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [ $\mu\text{g/L}$ , microgram per liter; MW, monitoring well]

**Table 6.** Summary matrix of the maximum probability of benzene hexachloride in groundwater exceeding various regulatory compliance goals at selected site locations, Oatland Island, Georgia.

[BHC, benzene hexachloride; Georgia EPD, Georgia Department of Natural Resources, Environmental Protection Division;  $\mu\text{g/L}$ , microgram per liter; MW, monitoring well; RRS, risk reduction standard; —, not available]

Georgia EPD regulatory compliance goal <sup>1</sup> ( $\mu\text{g/L}$ )		Maximum probability (percent) of BHC in groundwater exceeding compliance goals at selected locations (time period of maximum probability) <sup>2</sup>		
		MW-20 (2030)	MW-23 (2045)	Wetlands (2065)
Total BHC	Detection Limit: 0.044	89	45	13
Alpha-BHC	Type I RRS: 0.006 ( <sup>3</sup> 0.044)	96	80	24
	Type II RRS: 0.028 ( <sup>3</sup> 0.044)	72	49	14
Beta-BHC	Type I RRS: 0.02 ( <sup>3</sup> 0.044)	93	50	16
	Type II RRS: 0.097	83	36	8
Delta-BHC	Type I RRS: —	—	—	—
	Type II RRS: 0.66	60	18	4
Gamma-BHC	Type I RRS: 0.2	76	28	6
	Type II RRS: 0.66	60	18	4

<sup>1</sup> Georgia EPD Type I RRSs are defined in Georgia Rule 391-3-19.07 (Georgia EPD 1993). Georgia EPD Type II RRSs were calculated by a consultant using site-specific reasonable maximum exposure scenarios as defined in Georgia Rule 391-3-19.07(7) (MWH 2005, Georgia EPD 1993).

<sup>2</sup> From probabilistic results for a 300-year simulated time period (1970–2270). The selected site locations are listed in order of proximity to the source area: MW-20 is located approximately 300 ft downgradient of the source, MW-23 is 500 ft downgradient of the source, and the wetlands are 800 ft downgradient of the source.

<sup>3</sup> The detection limit for BHC (0.044  $\mu\text{g/L}$ ) is the default compliance standard if the Type I or Type II RRS is below the detection limit (Georgia EPD 1993), as defined in Georgia Rule 391-3-19-.07, Appendix II, Table 1 (Georgia EPD 1993).

MW-23 (Figures 12 and 13). Future groundwater monitoring results for MW-23 can be used both to evaluate actual contaminant plume migration and to gauge the conservatism of the model. If total BHC is consistently not detected at MW-23 during future monitoring events, then the analytical model is likely more conservative than actual site conditions, and model estimates for wetlands impact (maximum probability of 13% that total BHC will exceed the detection limit) may be too high. If BHC is detected in MW-23 during future sampling events, the MW-23 probabilistic type curves (Figures 13 and D13) provide a useful framework for interpreting the results. If sampling results at MW-23 consistently remain within model predictions, installation of an additional monitoring well to facilitate monitoring the leading edge (maximum travel distance) of the contaminant plume may be appropriate. Model simulations of the contaminant plume indicate a maximum travel distance of 40–60 feet beyond Barley Drive. On the basis of these results, the additional well should be located 100 feet beyond Barley drive, along the centerline of the plume.

Measured site sampling results during 2000–2005 indicated detectable concentrations of BHC in MW-17 and MW-20. Future sampling results for these monitoring wells can be compared with the probabilistic type curves developed for these wells (Figures D9 through D12) to evaluate the migration of BHC in groundwater over time. If the contaminant source was effectively eliminated during remediation during 2000, total BHC concentration in groundwater should decrease over time at MW-17 and MW-20. Given the physical and chemical properties of BHC, some contamination may be present in the source area in the form of residual DNAPL in the saturated zone (Pankow and Cherry 1996, USEPA 2003). If residual DNAPL is present, it will act as an ongoing source, and BHC concentrations in groundwater at MW-17 and MW-20 could remain relatively constant, with little or no decrease, until the residual DNAPL source is depleted. Deterministic model simulations for a constant source, which could represent the upper bound for a residual DNAPL scenario, indicate that the BHC plume stabilizes at 560 feet (60 feet beyond Barley Drive) after approximately 120 years (Figure 8F). Future groundwater monitoring results could be compared with the eliminated source and constant source simulation results shown in Figures 8 and 9 to assess the likelihood of residual DNAPL at the site.

The Georgia EPD Type II RRS for the beta-, delta-, and gamma-BHC isomers are 0.097  $\mu\text{g/L}$ , 0.66  $\mu\text{g/L}$ , and 0.66  $\mu\text{g/L}$ , respectively. These compliance standards are higher than the detection limit of 0.044  $\mu\text{g/L}$  (the default compliance standard referenced on the probabilistic type curves). Therefore, it may be useful to interpret the type curves in terms of a single isomer and its corresponding regulatory compliance goal (Table 6). Interpreting the probabilistic type curves in terms of each individual isomer is valid because the physical and chemical properties and partition coefficients for three of the four isomers are similar, have been well-studied, and were used to describe contaminant transport for total BHC.

## Recommendations

Based on field data and simulation results presented herein, ATSDR recommends monitored natural attenuation of the pesticide contaminants in groundwater at Area A as an effective next step in meeting regulatory compliance and public health goals. The probabilistic type curves developed and described in this report for MW-17, MW-20, MW-23, and the wetlands (Appendix D) should be used to evaluate future groundwater monitoring results and guide environmental and regulatory decision-making for the site.

ATSDR's recommendations are based upon the following:

- Source remediation (excavation of containers and contaminated soil) was completed during 2000 (MWH 2001).
- In-situ treatment of source area groundwater contaminants was conducted during 2003–2004 (MWH 2003, MWH 2005).
- No immediate or future public health issues are associated with the site.
- Analytical modeling results for pesticide migration indicate the probability of exceeding the detection limit of 0.044  $\mu\text{g/L}$  total BHC in groundwater at the wetlands boundary increases from 1% during 2000 to a maximum of 13% during 2065 (Figures 12 and 14, Table 6).
- MW-23, located along the centerline of the plume, half-way between the former source area and the wetlands, has not yet been affected by the contaminants (no detections of BHC in eight groundwater monitoring events during 2000–2005). Model simulations predict a 12% to 45% probability that total BHC will affect MW-23 above the detection limit during 2000–2045 (Figures 12 and 13). Future groundwater monitoring results for MW-23 could be used to evaluate actual contaminant plume migration and to gauge the conservatism of the model.

## Limitations of Models and Analyses

The ACTS software is a screening-level tool that uses mathematical solutions to model groundwater and contaminant transport. Calibrating an analytical model is not as rigorous or exact as calibrating more mathematically complex numerical models that are based on fewer simplifying assumptions and limitations. In this case, as is common with many environmental analyses, the mass of contaminant released at the site is unknown. Calibration based on a contaminant mass balance is not possible. Instead, calibration of groundwater and contaminant transport is accomplished by “plume-matching,” which means that the model input parameters are adjusted iteratively to produce simulation results that match, as closely as possible, the contaminant concentrations measured at the site over time.

Much of the uncertainty and variability in model parameters used to describe the hydrogeology and contaminant characteristics are captured in the PDFs generated for eight model variants. Still, differences and heterogeneities in actual site conditions could affect groundwater and contaminant transport in ways that are not represented by the analytical model because of limitations and simplifications of the governing equations of groundwater flow and contaminant transport. The primary limitation is the assumption of a constant and uniform groundwater velocity in the longitudinal direction (along the x-axis of the model grid).

In all model simulations, it was conservatively assumed that biological and chemical degradation of site contaminants was negligible. If BHC (or other site contaminants) are undergoing biological or chemical transformation at the site, model results would overestimate the time required for BHC concentrations to decrease at MW-17 and MW-20. Additionally, the estimated probability of affecting the wetlands at any given compliance goal—the detection limit or other applicable Georgia EPD risk reduction standard—would likely decrease.

For probabilistic simulations, it was assumed that remediation activities during 2000 effectively eliminated contaminants in the unsaturated zone (down to the water table) and no residual DNAPL was present in the saturated zone below Area A. If residual DNAPL was (or is) present, model results would underestimate the time required for BHC contaminant concentrations to decrease at MW-17 and MW-20. If residual DNAPL is present, it will act as an ongoing source, and BHC concentrations in groundwater could remain relatively constant, with little or no decrease, until the residual DNAPL source is depleted (Figures 8 and 9). In the deterministic simulations, the maximum travel distance of the contaminant plume is slightly higher in the constant source scenario versus the eliminated source scenario (560 feet versus 540 feet, respectively). Residual DNAPL (similar to the constant source scenario) could likewise result in a longer contaminant plume length and a higher probability of affecting the wetlands.

The presence of residual DNAPL in the saturated zone below the source area could introduce scenarios for contaminant migration that are not represented by the current model. If, for example, the residual DNAPL below the source area migrates down to the confining clay layer at 15–20 ft bgs (Figures A2 and A3), it could travel along the surface of the clay layer away from the original source area and into another area of the site. The original mass of the contaminant source is unknown and the topography of the clay layer is unknown. The potential extent, transport, and fate of residual DNAPL at Area A, therefore, cannot be effectively measured or predicted.

## Summary and Conclusions

ATSDR used a publicly available analytical modeling software application, the analytical contaminant transport analysis system (ACTS), to examine the fate and transport

of organochlorine pesticides in shallow groundwater at the Oatland Island Education Center, Oatland Island, Georgia. Specific objectives included: (1) estimating the probability of affecting coastal wetlands located 800 ft downgradient of the pesticide source area, and (2) developing reference tools (probabilistic type curves) for evaluating future groundwater monitoring results at key site monitoring wells.

Deterministic (single-point) modeling results presented herein are in good agreement with measured data from the Oatland Island site (Table 4, Figures 4 and 5). Deterministic simulations using calibrated, single value input parameters indicate the contaminant plume will not affect the wetlands. Deterministic results indicate total BHC concentrations at the wetlands are below the BHC detection limit (0.044 µg/L) and below all relevant BHC reference values for regulatory and health guidance.

Probabilistic results derived by conducting a two-stage Monte Carlo analysis using 10,000 realizations for eight different input parameters indicate that the probability of exceeding the detection limit of 0.044 µg/L total BHC in groundwater at the wetlands boundary increases from 1% during 2000 to a maximum of 13% during 2065 (Figures 12 and 14). This represents an 87% confidence level that the wetlands will not be affected in the future by pesticide migration from Area A. For probabilistic simulations, it was assumed that source remediation activities during 2000 effectively eliminated source area contaminants. However, some residual contamination likely is still present in the source area in the form of residual DNAPL in the saturated zone. The presence of residual DNAPL could increase the horizontal extent of the BHC plume and increase the probability of BHC affecting the downgradient wetlands.

At MW-23, located along the centerline of the plume, halfway between the former source area and the wetlands, model results indicate a 12% to 45% probability that total BHC will be detected at levels above the detection limit during 2000–2045 (Figures 12 and 13). BHC was not detected at MW-23 in the eight sampling events conducted during 2000–2005. Long term monitoring at MW-23—and at other key monitoring wells within Area A—is a practical approach that enables evaluation of actual contaminant plume migration, comparison of groundwater monitoring results with model predictions, and timely decision-making for the site that will be protective of the wetlands.

The probabilistic type curves developed for the four site locations (MW-17, MW-20, MW-23, and the wetlands boundary) can be used to evaluate future groundwater monitoring results and guide environmental and regulatory decision-making for the Oatland Island site. The type curves for MW-17, MW-20, MW-23, and the wetlands are presented in terms of total BHC, but they also may be interpreted in terms of a single isomer and its corresponding regulatory compliance target (Table 6). This may be a useful approach given the different Georgia EPD Type II RRS compliance targets for individual isomers (0.044 µg/L for alpha-BHC, 0.097 µg/L for beta-BHC, 0.66 µg/L for delta-BHC, and 0.66 µg/L gamma-BHC).

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## **Appendix A. Selected Illustrations from Previous Reports**



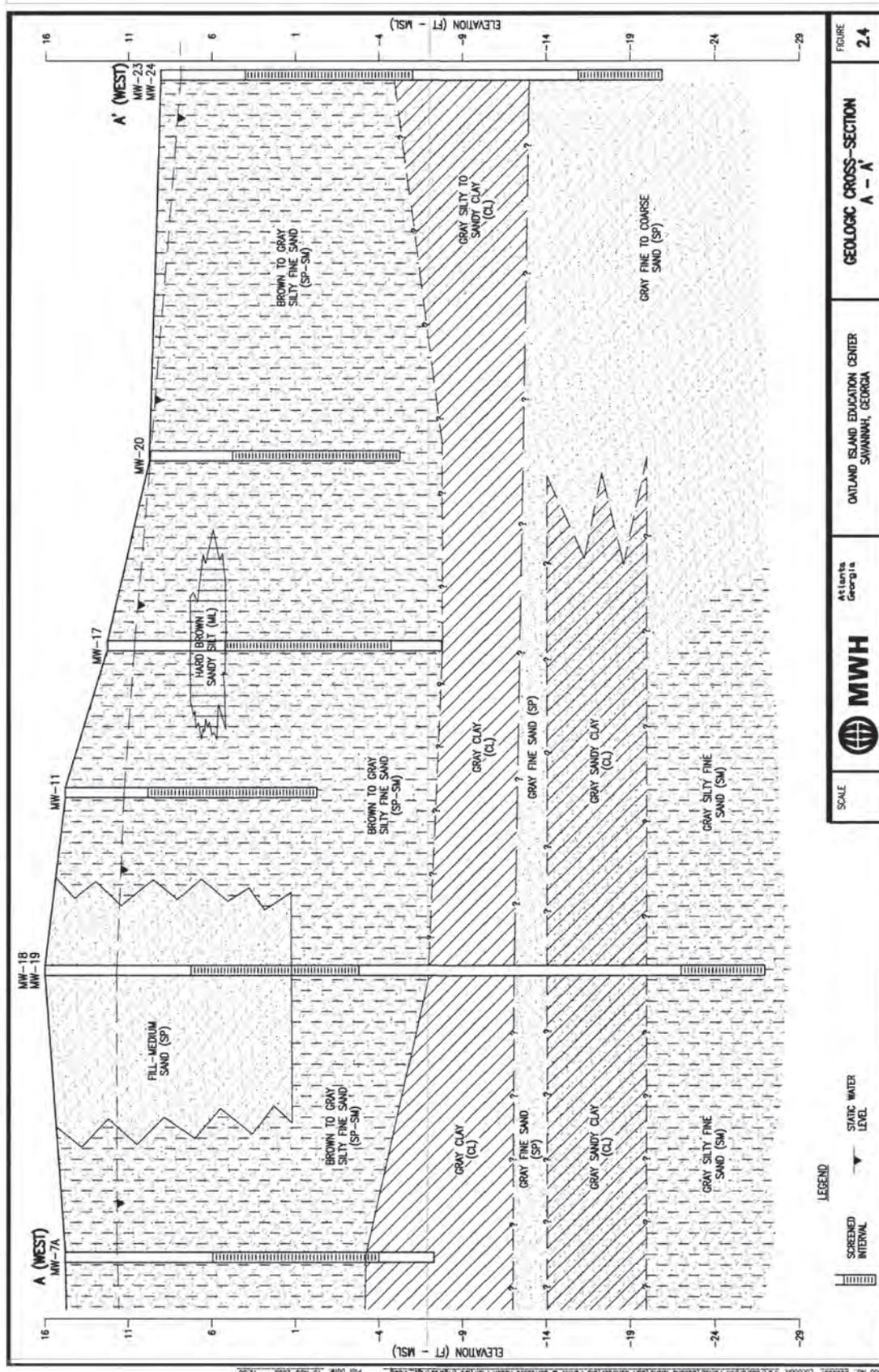


Figure A2. Geologic cross-section A-A' (MWH 2003, Figure 2.4).

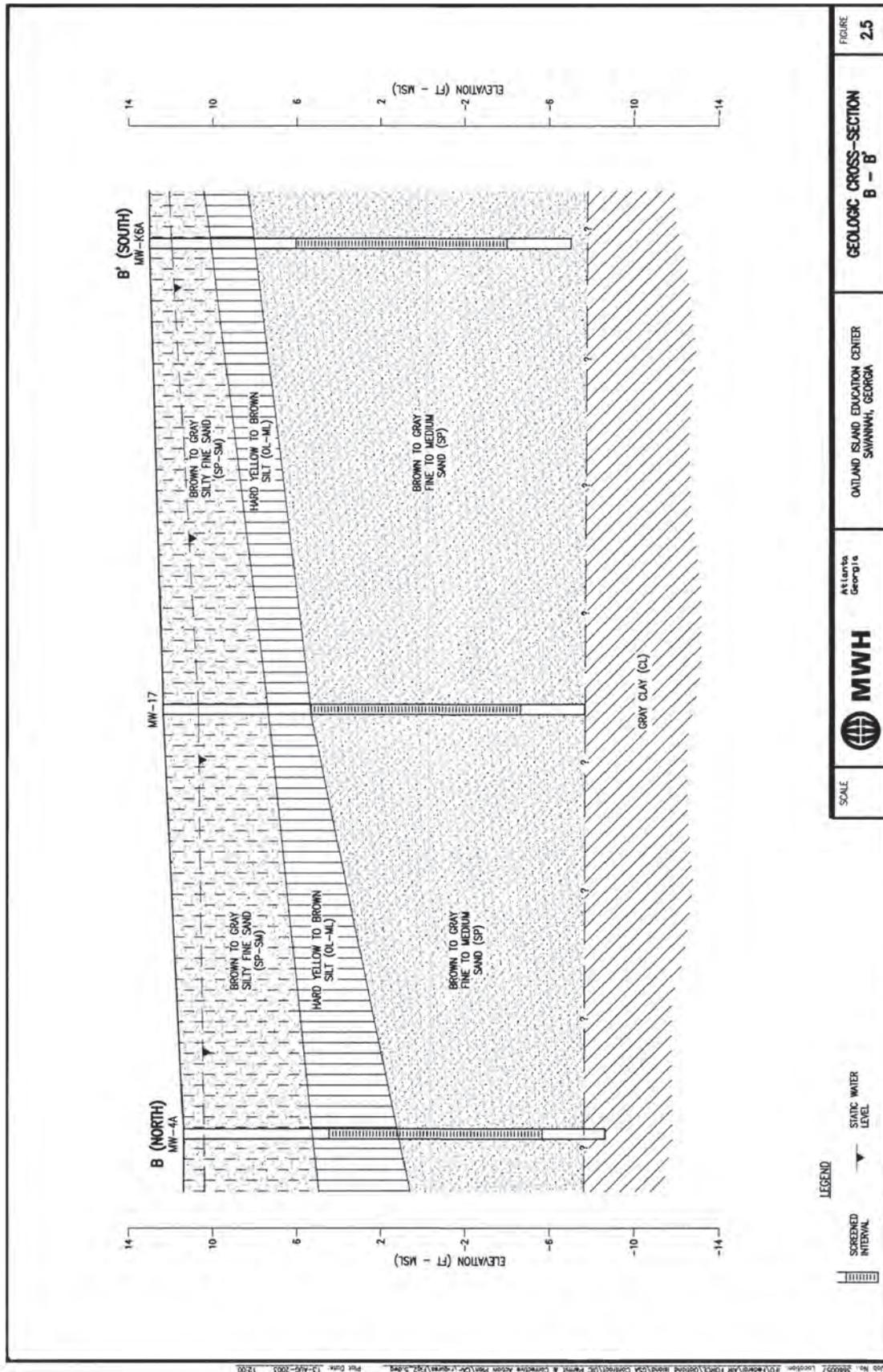


Figure A3. Geologic cross-section B-B' (MWH 2003, Figure 2.5).





## Appendix B. Summary of Research on Media and Contaminant Characteristics

Media and contaminant parameter values were obtained from a variety of sources, including existing site field data and analyses, available USGS information for the area, relevant literature articles, and other published references on groundwater and contaminant transport. These values are listed in Table B1. Abbreviations cited in Table B1 are listed below. References cited in Table B1 are listed in the report reference section on page 24.

### Table B1 abbreviations

ATSDR	Agency for Toxic Substances and Disease Registry
ca.	calculated
d	day
<i>ET</i>	evapotranspiration
$f_{oc}$	fraction of organic carbon in soil
ft	feet
g	gram
$K_{oc}$	soil organic carbon partition coefficient
$K_{ow}$	octanol-water partition coefficient
$K_D$	soil distribution coefficient or soil-water partition coefficient
L	liter
lb	pound
mg	milligram
mi	mile
mL	milliliter
MWH	Montgomery Watson Harza Americas, Inc.
<i>P</i>	precipitation
ppm	parts per million
<i>q</i>	recharge
$Q_s$	surface runoff
yr	year
USEPA	U.S. Environmental Protection Agency

**Table B1.** Hydrogeologic and contaminant transport parameter values, Oatland Island Education Center, Oatland Island, Georgia.

Property	Value	Units	Reference	Notes
Hydraulic conductivity ( <i>K</i> )	0.023, 0.096	ft/d	S&ME (1999b)	MWH ca. from site-specific rising head slug tests on monitoring wells G5.5 (Area A) and MWT+30 (Area B).
	7.973–9.496		MWH (2005)	MWH and ATSDR ca. from five rising head slug tests on site monitoring wells [MWH (2005)].
	52		ATSDR ca. from MWH (2001)	ATSDR ca. using site-specific tidal and monitoring well data from MWH 2001. (See Appendix C of this report)
	41		ATSDR ca. from S&ME (1999b)	ATSDR ca. using site-specific grain-size analysis data in Hazen’s approximation (Freeze & Cherry 1997).
	2–65		S&ME (1986), Clarke et al. (1990)	S&ME ca. from grain-size analysis data for adjacent Skidaway Island, Georgia (S&ME 1986).
	0.057–57		Domenico and Schwartz (1998)	Range of <i>K</i> values for fine sand.
Hydraulic gradient ( <i>i</i> )	31.7	ft/mi	S&ME (1999b)	Average gradient between monitoring wells MW-5 and K-6 (Area A).
	50.16		ATSDR ca. from MWH (2003), Figure 4.7	Gradient between MW-11 and MW-23 (Area A).
	27.46		MWH (2005)	Gradient measured in shallow groundwater across distal portion of dissolved BHC plume.
	35.38		ATSDR ca.	ATSDR ca. geometric mean of the three available historical gradients.
Porosity of soil ( $\theta$ )	0.3	—	S&ME (1999b)	Based on site-specific grain-size analyses of subsurface soil indicating 90% fine sand.
	0.3–0.5		Freeze and Cherry (1997)	Typical range for nonindurated sands.
	0.26–0.53		Morris and Johnson (1967)	Typical range for fine sands.
Recharge to surficial aquifer ( <i>q</i> )				$q = P - Q_s - ET$
Precipitation ( <i>P</i> )	30.46–65.95	in/yr	Southeast Regional Climate Center online data	Range of annual means (1948–1977) for Savannah Beach, GA ( <a href="http://www.dnr.state.sc.us/climate/sercc/climateinfo/historical/historical.html">http://www.dnr.state.sc.us/climate/sercc/climateinfo/historical/historical.html</a> ).
	50		National Weather Service data cited in Priest (2004)	Mean annual precipitation (1971–2000).
Surface runoff ( $Q_s$ )	10		Krause and Randolph (1989), Priest (2004)	
Evapotranspiration ( <i>ET</i> )	34–35		Krause and Randolph (1989), Priest (2004)	
Stream baseflow	5.41–7.16		Priest (2004)	Mean annual stream baseflow estimates for a number of sites in coastal Georgia where the surficial aquifer is the outcropping unit.

**Table B1.** Hydrogeologic and contaminant transport parameter values, Oatland Island Education Center, Oatland Island, Georgia. —Continued

Property	Value	Units	Reference	Notes
Bulk density of soil ( $\rho_b$ )	96.8	lb/ft <sup>3</sup>	Morris and Johnson (1967)	Mean value for fine sand.
	70.5–124		Morris and Johnson (1967)	Typical range for fine sand.
Longitudinal dispersivity ( $\alpha_L$ )	25–50	ft	5%–10% of aquifer length, from Fetter (1993), Gelhar et al. (1992); site-specific aquifer length (contaminant travel distance) estimated as 500 ft	Dispersivity is a function of the scale of the study. As the flow path increases, contaminant dispersion (mechanical mixing effects) will likewise increase (Fetter 1993).
Ratio of dispersivities ( $\alpha_L/\alpha_T$ )	10–100		Fetter (1993); Gelhar et al. (1992)	
Solubility in water ( <i>S</i> )				
BHC isomers	insoluble–69.5	mg/L	Range of six values listed in ATSDR (2005b) [Clayton & Clayton (1981), Kurihara et al. (1973), Hazardous Substances Data Bank (HSDB) (2003), Hollifield (1979)]	
gamma-BHC	6.6, 7, 11		US Department of Agriculture (USDA) (2004)	
gamma-BHC	0.14–15.2		Range of 63 values listed in Mackay et al. (1997)	
gamma-BHC	6.8		Mean of 63 values listed in Mackay et al. (1997)	
aldrin, dieldrin	0.011, 0.11		Bus and Leber (2001) [cited in ATSDR (2005b)]	
Retardation coefficient ( <i>R</i> )				
				$R = 1 + (\rho_b / \theta) K_D$ ; $\rho_b = 1.6$ g/mL; $\theta = 0.4$ ; $K_D = f_{OC} \times K_{OC}$ , where $f_{OC} = 0.001$ , $K_{OC}$ determined experimentally or calculated from $K_{OC} - K_{OW}$ relations
alpha-BHC	3.5–15.9	—	<i>R</i> ca. using range of three $K_{OC}$ values cited from various sources in ATSDR (2005b)	$K_{OC}$ determined experimentally
	8.0		<i>R</i> ca. using geometric mean of 12 $K_{OC}$ values from USEPA (1996)	$K_{OC}$ determined experimentally
	5.1–12.6		<i>R</i> ca. using range of 12 $K_{OC}$ values from USEPA (1996)	$K_{OC}$ determined experimentally
	12.1–22.5		MWH (2005); <i>R</i> ca. using three $K_{OW}$ values	$K_{OW}$ determined experimentally, $K_{OC}$ ca. from three different empirical $K_{OC} - K_{OW}$ relationships
beta-BHC	15.9	—	<i>R</i> ca. using $K_{OC}$ values from Ripping (1972) [cited in ATSDR (2005b)]	$K_{OC}$ determined experimentally
	9.6		<i>R</i> ca. using geometric mean of 14 $K_{OC}$ values from USEPA (1996)	$K_{OC}$ determined experimentally
	5.6–15.3		<i>R</i> ca. using range of 14 $K_{OC}$ values from USEPA (1996)	$K_{OC}$ determined experimentally
	11.8–21.5		MWH (2005); <i>R</i> ca. using three $K_{OW}$ values	$K_{OW}$ determined experimentally, $K_{OC}$ ca. from three different empirical $K_{OC} - K_{OW}$ relationships

**Appendix B. Summary of Research on Media and Contaminant Characteristics**

**Table B1.** Hydrogeologic and contaminant transport parameter values, Oatland Island Education Center, Oatland Island, Georgia.  
—Continued

Property	Value	Units	Reference	Notes
delta-BHC	26.2	—	<i>R</i> ca. using $K_{oc}$ values from Weiss (1986) [cited in ATSDR (2005b)]	$K_{oc}$ determined experimentally
	19.1–49.1		MWH (2005); <i>R</i> ca. using $K_{ow}$ value	$K_{ow}$ determined experimentally, $K_{oc}$ ca. from three different empirical $K_{oc} - K_{ow}$ relationships
gamma-BHC	3.6–15.9	—	<i>R</i> ca. using range of six $K_{oc}$ values cited from various sources in ATSDR (2005b)	$K_{oc}$ determined experimentally
	6.4		<i>R</i> ca. using geometric mean of 65 $K_{oc}$ values from USEPA (1996)	$K_{oc}$ determined experimentally
	3.9–14.0		<i>R</i> ca. using range of 65 $K_{oc}$ values from USEPA (1996)	$K_{oc}$ determined experimentally
	11.1–18.8		MWH (2005); <i>R</i> ca. using three $K_{ow}$ values	$K_{ow}$ determined experimentally, $K_{oc}$ ca. from three different empirical $K_{oc} - K_{ow}$ relationships
aldrin	187,095	—	<i>R</i> ca. using $K_{oc}$ values from Briggs (1981) [cited in ATSDR (2005b)]	$K_{oc}$ determined experimentally
	196		<i>R</i> ca. using geometric mean of two $K_{oc}$ values from USEPA (1996)	$K_{oc}$ determined experimentally
dieldrin	18,710	—	<i>R</i> ca. using $K_{oc}$ values from Briggs (1981) [cited in ATSDR (2005b)]	$K_{oc}$ determined experimentally
	103		<i>R</i> ca. using geometric mean of three $K_{oc}$ values from USEPA (1996)	$K_{oc}$ determined experimentally

## **Appendix C. Calculating Site-Specific Aquifer Characteristics Using Tidal Information**

Water level fluctuations in an inland monitoring well influenced by ocean tides can be used to calculate site-specific transmissivity and storativity relationships as follows:

$$t_L = x [(t_o S)/(4\pi T)]^{1/2} \quad (1)$$

where

- $t_L$  is the time lag of a maximum or minimum water level measured in the monitoring well,
- $x$  is the distance inland to the monitoring well,
- $t_o$  is the tidal period,
- $S$  is the storativity of the aquifer, and
- $T$  is the transmissivity of the aquifer.

The development of equation 1 is outlined in several groundwater hydrogeology texts (Fetter 1988, Todd 1980). Although primarily applicable to confined aquifers, this equation can also be applied to unconfined aquifers if the tidal fluctuations are small compared to the aquifer saturated zone thickness (Fetter 1988, Todd 1980).

On October 11, 2000, MWH conducted a 12-hour (hr) tidal study at the Oatland Island site to quantify the influence of ocean tides on measured water levels in site monitoring wells (MWH 2001). The following information was documented during the study:

- Fluctuation at the Richardson Creek dock (Figure C1) was 7 ft over 12 hrs.
- Low tide at the dock occurred at 7:25 am; high tide occurred approximately 6 hrs later.
- Water level measurements collected at 2-hr intervals for 10 monitoring wells in Area A indicated a maximum fluctuation of 0.04 ft.
- Water level measurements collected at 2-hr intervals for 6 monitoring wells in Area B indicated a maximum fluctuation of 0.12 ft.

Only Area B monitoring well data were selected for calculations because Area A wells may be influenced by two different reaches of Richardson Creek (Figure C1). Additionally, more water level measurements are available for Area B

wells to allow confirmation of the time lag to minimum water level (in the monitoring wells) associated with low tide at Richardson Creek. The MWH tidal study data (Table C1) and tidal data for the closest National Oceanic and Atmospheric Association tidal monitoring station (Fort Pulaski station) are presented graphically in Figure C2.

The surficial aquifer thickness in Areas A and B is approximately 14 ft. The observed water level fluctuations were small compared to the aquifer thickness (0.1–0.12 ft; less than 1% of the aquifer thickness), therefore equation 1 can be used for the unconfined aquifer at the site (Fetter 1988, Todd 1980).

Rearranging equation 1 and solving for T/S,

$$T/S = (t_o/4\pi)(t_L/x)^2 \quad (2)$$

where

- $t_o$  equals 6 hrs (tidal period from measurements at Richardson Creek dock),
- $t_L$  equals 11.3 hrs (Table C1, Figure C2,)
- $x$  equals 900 ft for Area B monitoring wells,

therefore

$$T/S \text{ equals } 3,029 \text{ ft}^2/\text{hr}.$$

Storativity (S) values for unconfined aquifers (specific yield) typically range from 0.01 to 0.3, while storativity values for confined aquifers range from 0.005 to 0.00005 (Freeze and Cherry 1979). Because the surficial aquifer at Oatland Island is generally unconfined with some laterally discontinuous clay lenses, a storativity value of 0.01 was assumed for initial calculations. Transmissivity (T) is a function of hydraulic conductivity (K) and aquifer thickness (b):

$$T = bK \quad (3)$$

Combining equations (2) and (3) and solving for K,

- $bK/S$  equals 3,029 ft<sup>2</sup>/hr
- $K$  equals (3,029 ft<sup>2</sup>/hr)(S/b)
- $K$  equals (3,029 ft<sup>2</sup>/hr)[(0.01)/(14 ft)]
- $K$  equals 2.16 ft/hr or 52 ft/day.

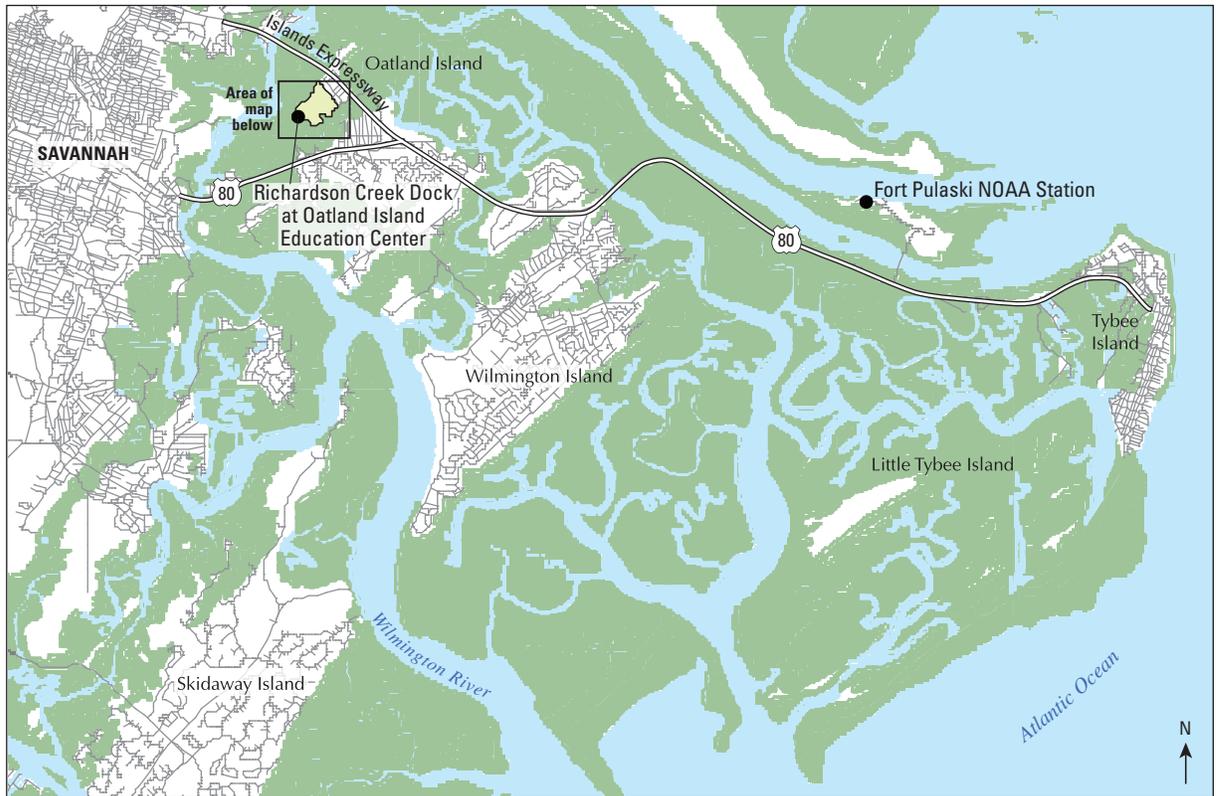
Summary of calculations:

For  $S=0.01$   $K$  is calculated as 52 ft/day.

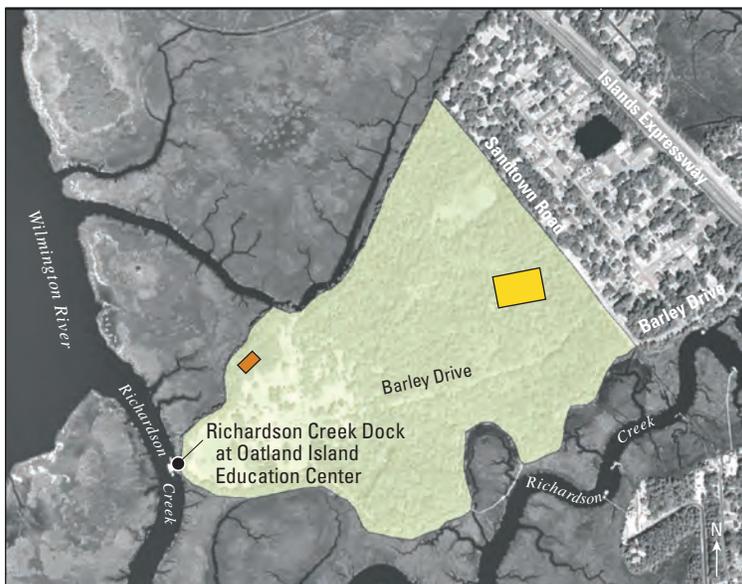
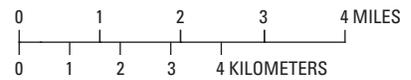
**Table C1.** Summary of 12-hr tidal study data, Oatland Island, Georgia (from MWH, 2001).

[ $x$ , distance from Richardson Creek dock; ft, feet;  $t_L$ , time lag; hr, hour; —, insufficient data collected to determine this information]

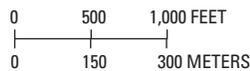
Location (monitoring well designation)	Distance from Richardson Creek dock ( $x$ , ft)	Maximum water-level fluctuation (ft)	Time of minimum water-level occurrence	Time lag ( $t_L$ , hrs)
Richardson Dock	0	7	7:25 AM	0
Area A (MW-4)	2,500	0.04	—	—
Area A (K-6)	2,500	0.04	—	—
Area B (MWT+40D25)	900	0.12	6:45 PM	11.3
Area B (MWT+70D25)	900	0.11	6:45 PM	11.3



Base modified from Montgomery Watson Harza, 2005  
 Wetlands boundaries based on National Wetlands Inventory data  
 from U.S. Fish & Wildlife Service, 2004



1999 aerial photograph courtesy of U.S. Geological Survey  
 through Terraserver USA

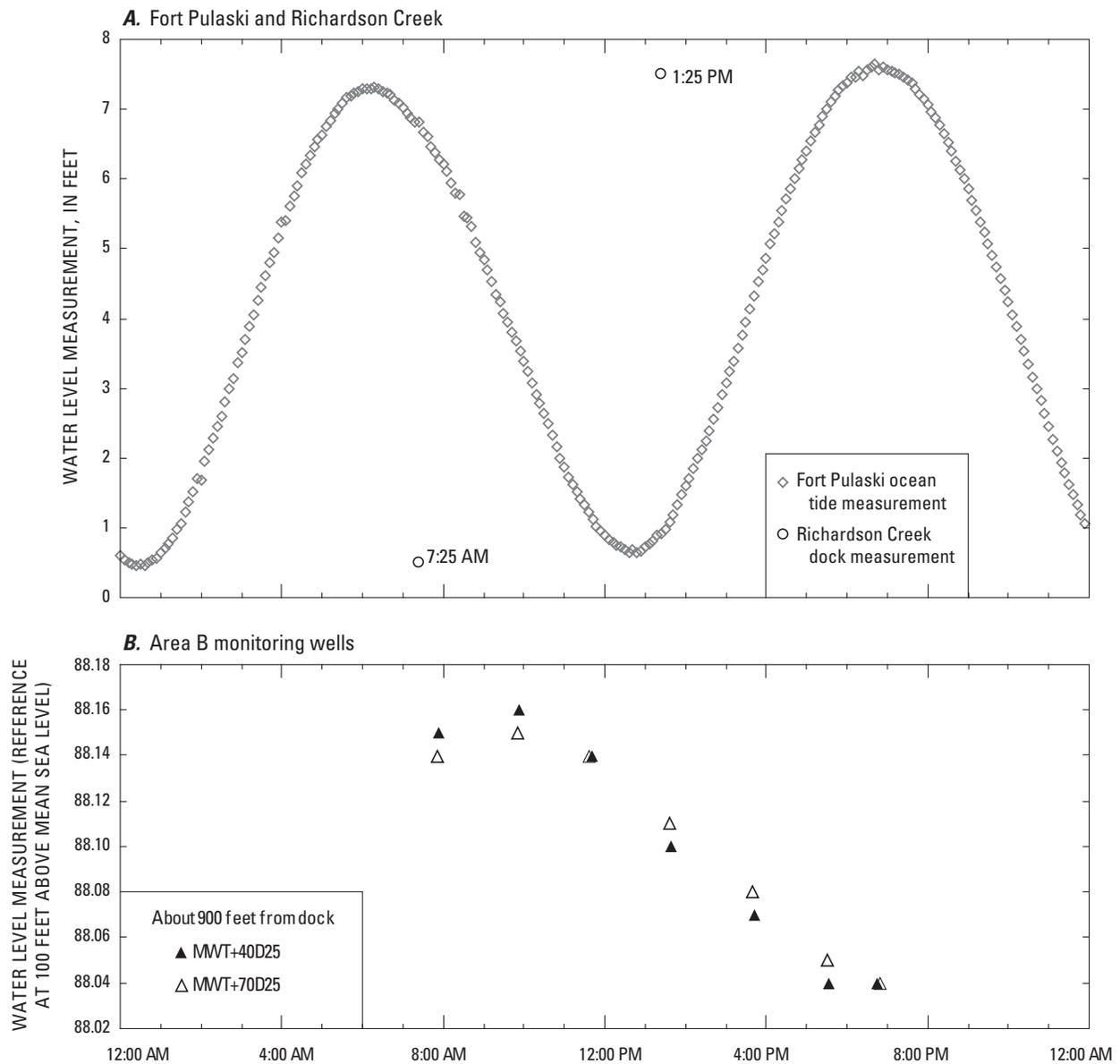


**EXPLANATION**

- Oatland Island Education Center
- Area A
- Area B
- Wetland
- Water
- Tidal study reference point



**Figure C1.** Tidal study reference points, Oatland Island Education Center, Oatland Island, Chatham County, Georgia. [NOAA, National Oceanic and Atmospheric Administration]



**Figure C2.** Tidal data and monitoring well measurements for Oatland Island, Georgia: (A) Fort Pulaski NOAA station ocean tide and Richardson Creek tide measurements and (B) water level measurements for Area B monitoring wells (MWH 2001). [NOAA, National Oceanographic and Atmospheric Administration; MW, monitoring well]

## Appendix D. Probabilistic (Monte Carlo) Simulations

A two-stage Monte Carlo simulation was used to conduct probabilistic analyses of contaminant transport. In stage 1, probability density functions (PDFs) were developed with the Analytical contaminant analysis system (ACTS) version 7.8 software<sup>1</sup> for eight input parameters—referred to as variants—using Monte Carlo simulation (Table 5). In stage 2, values from each PDF were randomly selected as input parameters for the fate and transport model within ACTS to generate contaminant concentrations in groundwater for a specific time and location at the site (for example, selected monitoring wells or the wetlands boundary).

### Stage 1: PDFs for Eight Model Variants

The PDFs generated within ACTS for each of the eight variants are shown in Figures D1 through D4. Descriptive statistics for the PDFs are summarized in Table 5. Each PDF consists of a range of 10,000 random values (or realizations) that characterizes the variability and uncertainty of the input parameter. Within ACTS, the type (shape) and the descriptive statistics (mean, minimum, maximum, variance) for each PDF were specified on the basis of the available site-specific field data and applicable literature references (Appendix B).

### Stage 2: Fate and Transport Modeling

Using the eight PDFs as input, 10,000 Monte Carlo realizations were run within ACTS to generate a range of possible contaminant concentrations in groundwater for a specific time and location of interest. The probabilistic simulations were conducted using the eliminated source scenario (Table 3).

The results of a probabilistic analysis of total BHC concentration at a given location and time can be expressed as a histogram/frequency distribution, cumulative frequency distribution, probability distribution, cumulative probability distribution, or a complementary cumulative probability distribution. Simulation results for 10,000 Monte Carlo realizations at MW-20 during 2003 (simulation time of 33 years) are shown in terms of each of these plot types in Figures D5 through D8:

- Figure D5 is a distribution representing the frequency of occurrence for a range of simulated total BHC concentration values.
- Figure D6 is a distribution representing the probability of occurrence (frequency of occurrence divided by the number of realizations) for a range of concentration values.

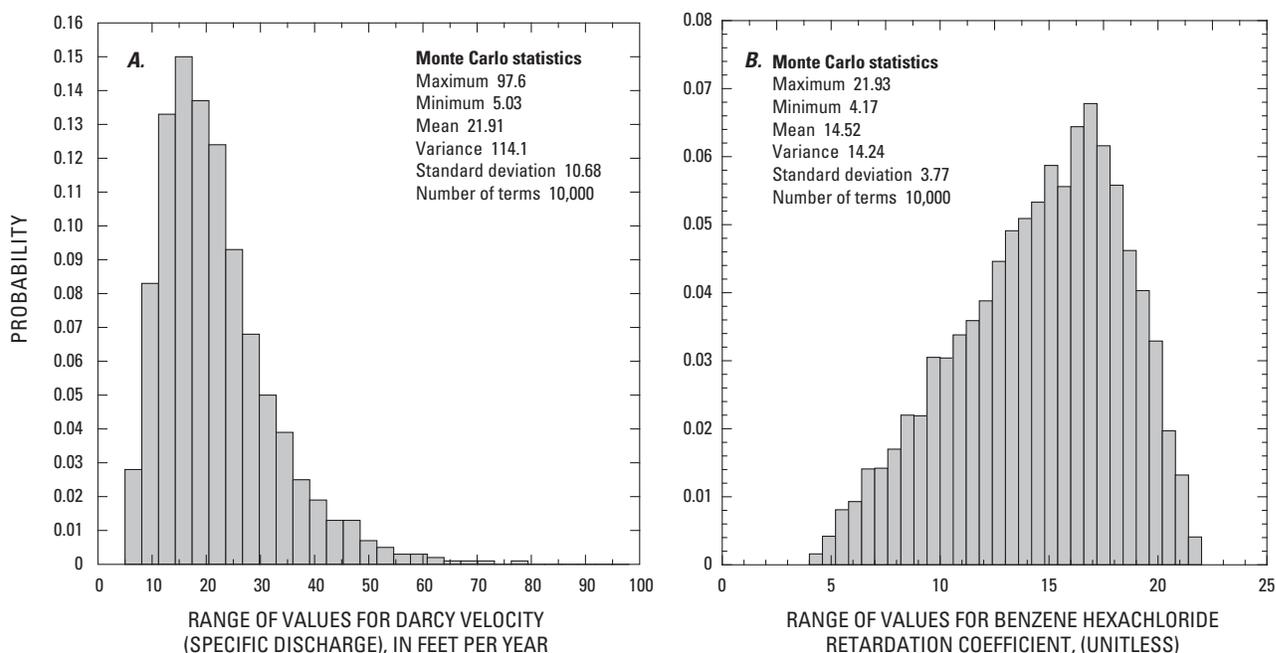
- Figure D7 is a distribution representing the cumulative probability of occurrence (the cumulative probability that simulated concentrations will be below a selected concentration) for a range of simulated BHC concentration values. By definition, the cumulative probability function sums to a value of 1 (Figure D7A). In Figure D7B, values do not sum to 1 because only part of the graph is shown to highlight values of interest.
- Figure D8 is a distribution representing the complementary cumulative probability of occurrence (the cumulative probability that simulated concentrations will exceed a selected concentration) for a range of simulated BHC concentration values.

During 2003, measurements of total BHC concentrations at MW-20 ranged from 0.8 to 2.2  $\mu\text{g/L}$ . Probabilistic simulation results for a median value of 1.5  $\mu\text{g/L}$  total BHC are labeled in the cumulative probability and complementary cumulative probability plots shown in Figures D7 and D8, respectively, to demonstrate the practical interpretation of these plots. From Figure D7, the probability that total BHC would be equal to or less than 1.5  $\mu\text{g/L}$  is estimated to be 66%. The probability that total BHC would exceed 1.5  $\mu\text{g/L}$  would be 1 minus the cumulative probability function (1 minus 66%), which is shown in Figure D8 as 34%.

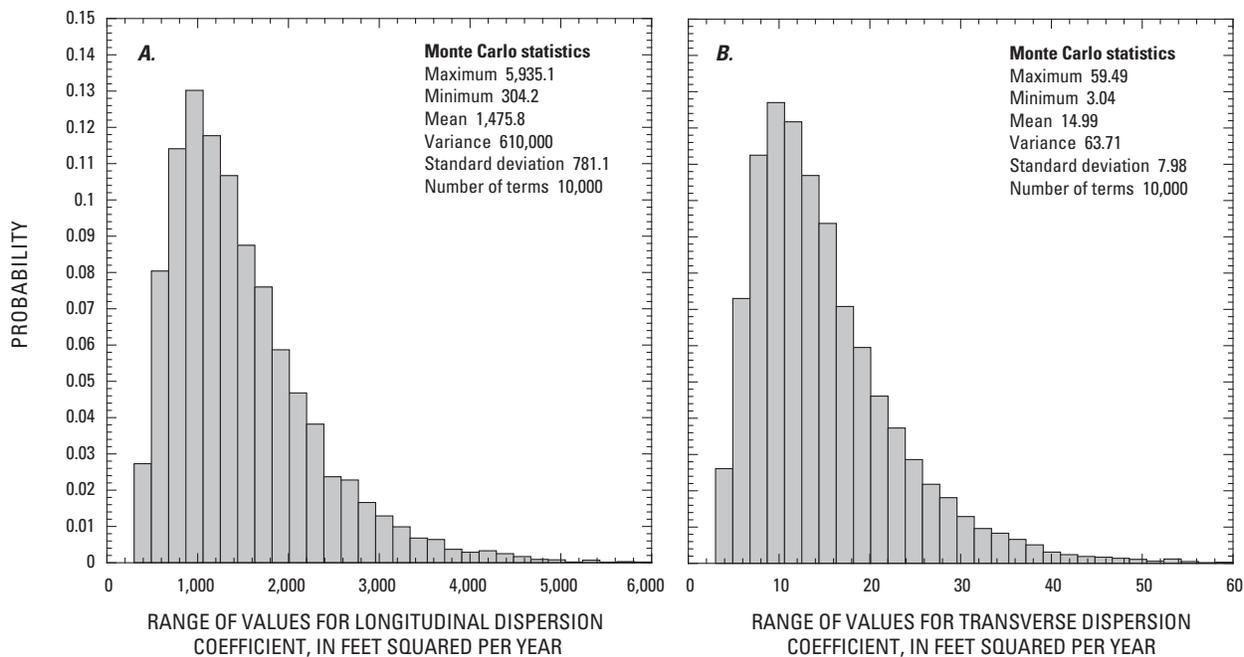
Another way to represent the probabilistic analysis is to aggregate model simulation results for multiple time periods at a selected location to form a series of probabilistic “type curves” on a single plot to illustrate the probability of exceeding certain compliance goals, such as a detection limit or regulatory action level, over time. The probabilistic type curves were developed by running ACTS in Monte Carlo mode for a single location and time period, importing the results (10,000 values of total BHC concentration) into a spreadsheet, updating and running ACTS again for the same location and new time period, importing the new results into the spreadsheet, and repeating this process for a number of different time periods.

Probabilistic type curves developed for each of four site locations of interest (MW-17, MW-20, MW-23, and the wetlands) are included in Figures D9 through D16. Two sets of type curves were developed for each location: one for time periods before and one for time periods after the contaminant plume reaches its maximum probability of exceeding the BHC detection limit at that location. The probability of exceeding a selected concentration at a given time period is determined from the type curves by selecting the concentration of interest on the x-axis, drawing a vertical line that intersects the selected simulation time curve, and drawing a horizontal line that intersects the y-axis (probability of exceedance).

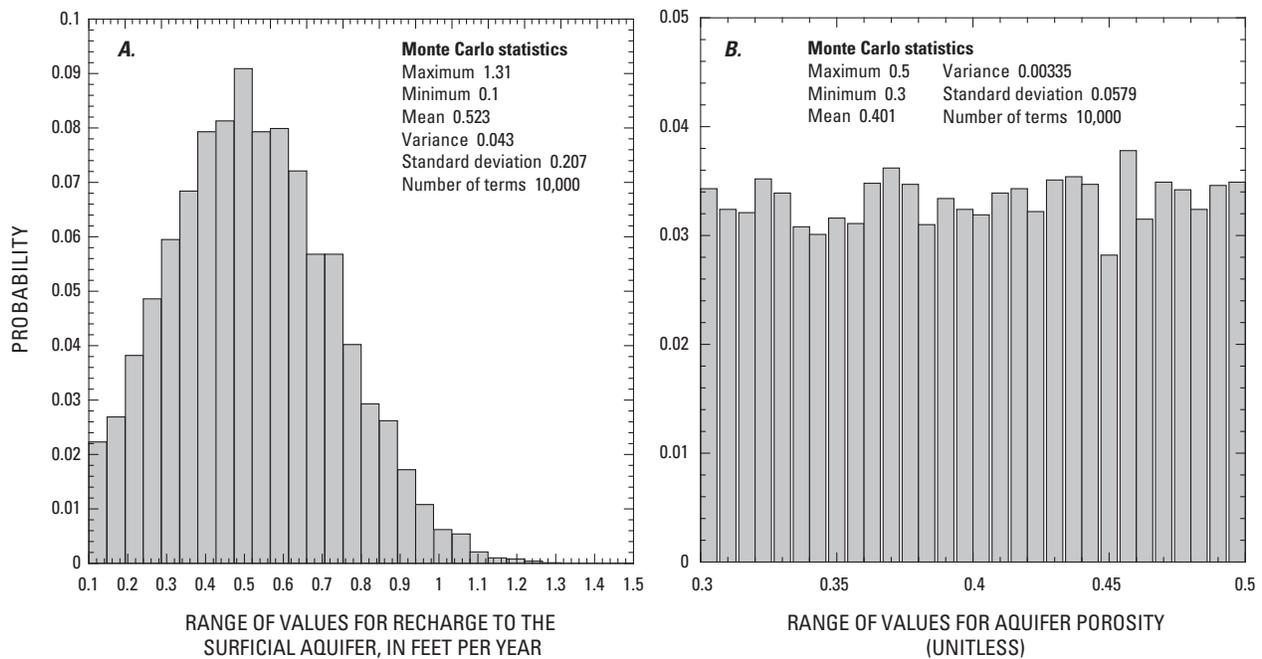
<sup>1</sup>ACTS software is free to the public. It can be downloaded from the Georgia Institute of Technology, School of Civil and Environmental Engineering, Multimedia Environmental Simulations Laboratory Web site: <http://mesl.ce.gatech.edu/>.



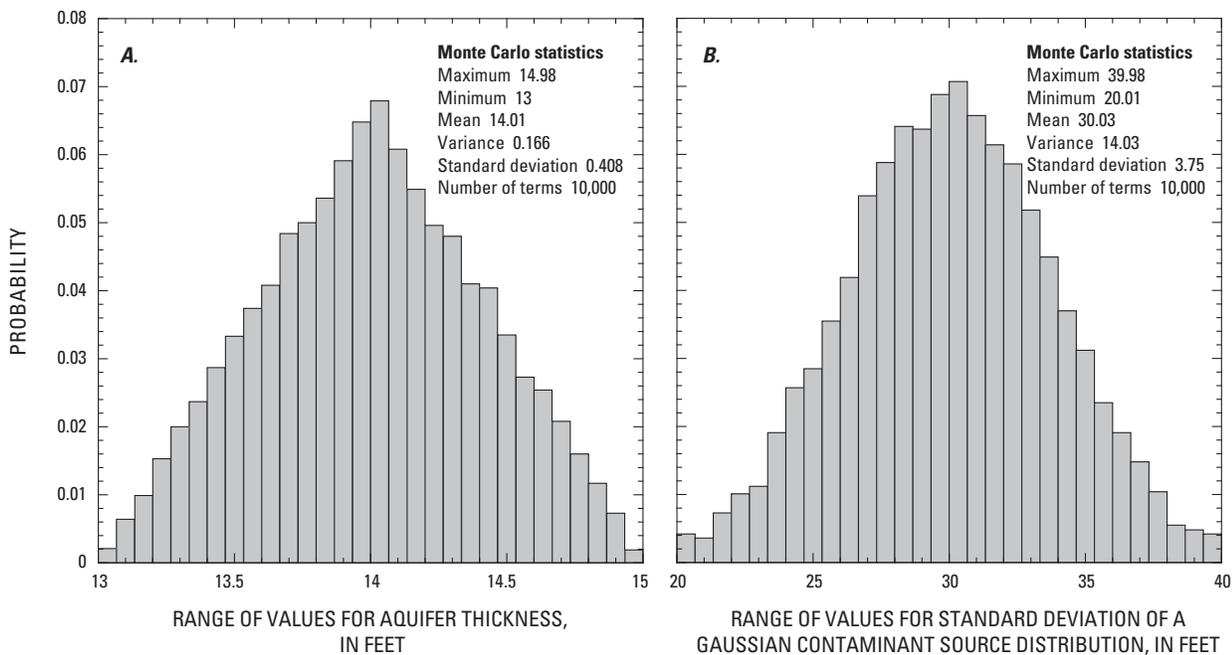
**Figure D1.** Probability density generated using 10,000 Monte Carlo realizations: (A) Darcy velocity, in feet per year, and (B) retardation coefficient (unitless) for benzene hexachloride.



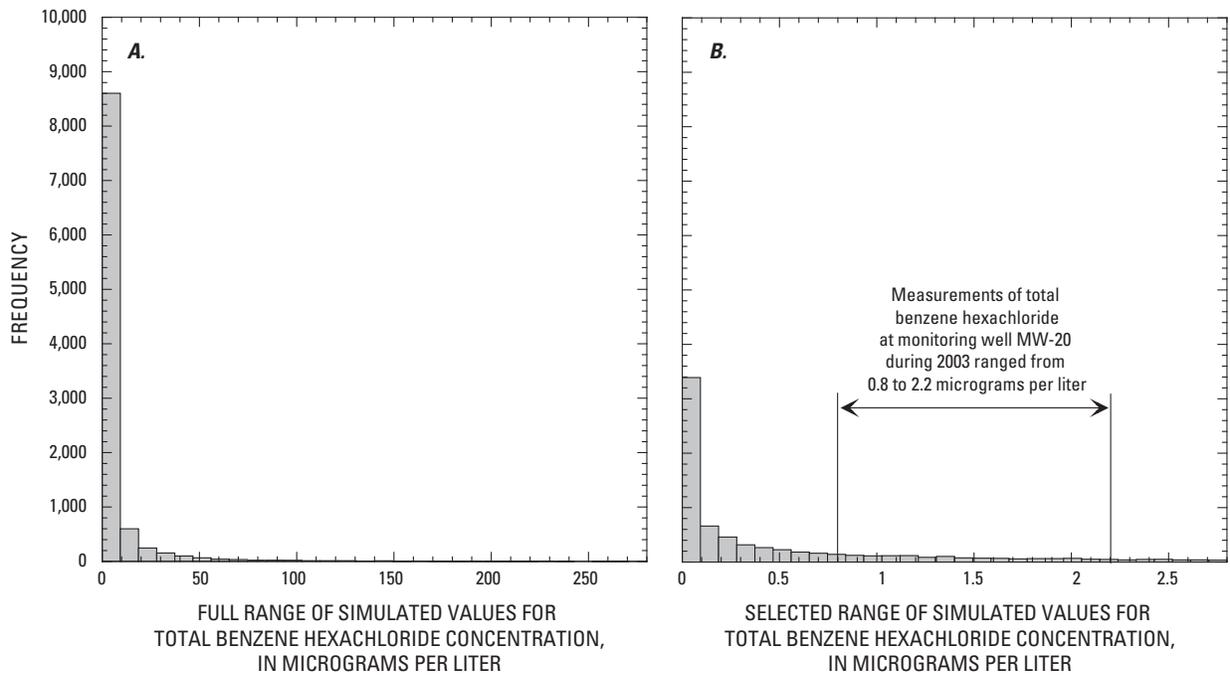
**Figure D2.** Probability density generated using 10,000 Monte Carlo realizations: (A) longitudinal and (B) transverse dispersion coefficients, in feet squared per year.



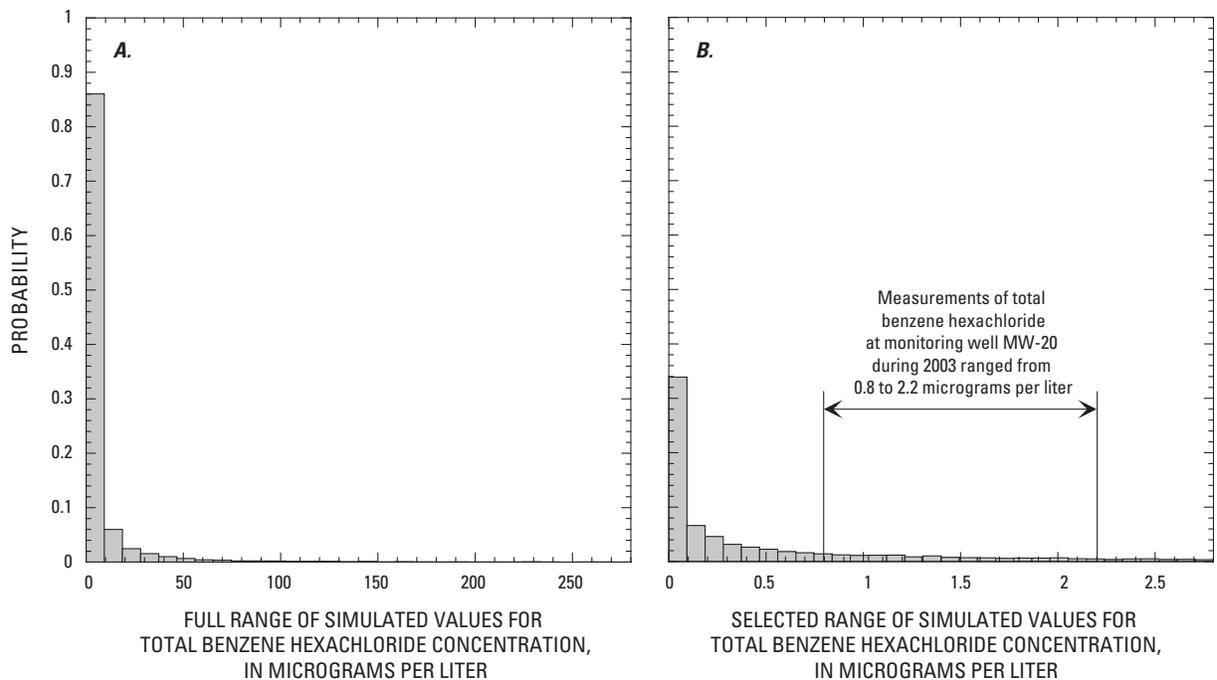
**Figure D3.** Probability density generated using 10,000 Monte Carlo realizations: (A) infiltration (recharge) to the surficial aquifer, in feet per year, and (B) aquifer porosity (unitless).



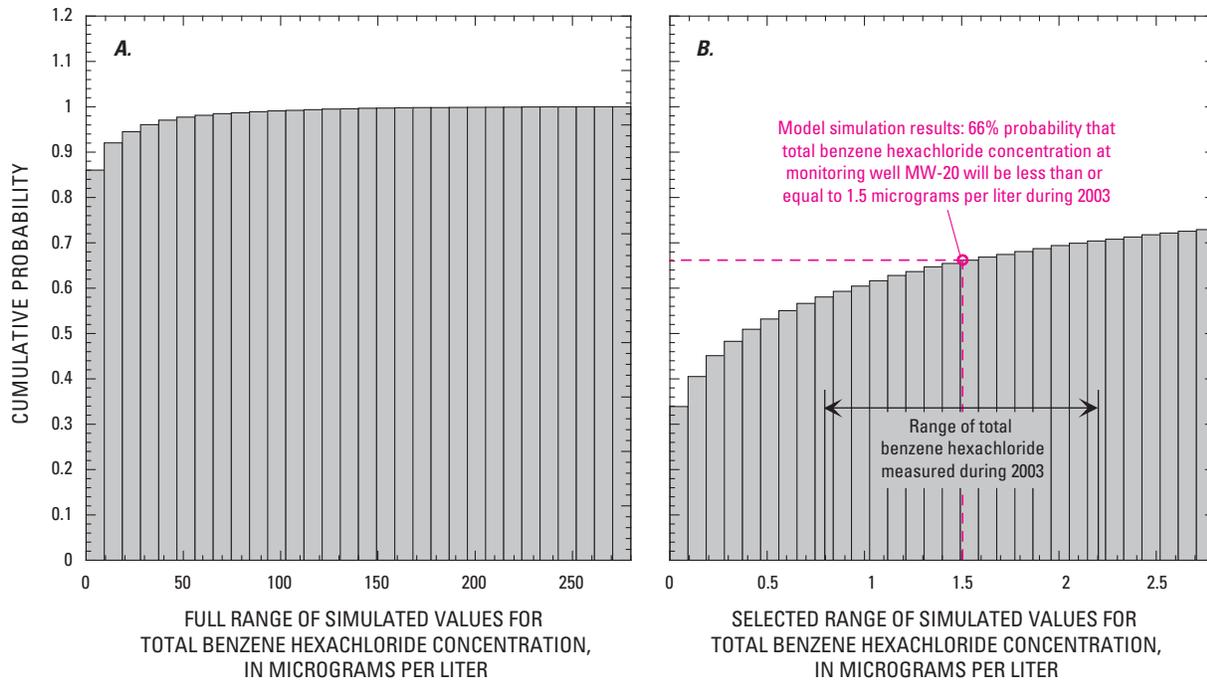
**Figure D4.** Probability density generated using 10,000 Monte Carlo realizations: (A) aquifer thickness, in feet, and (B) standard deviation of the source width ( $\sigma_w$ ), in feet.



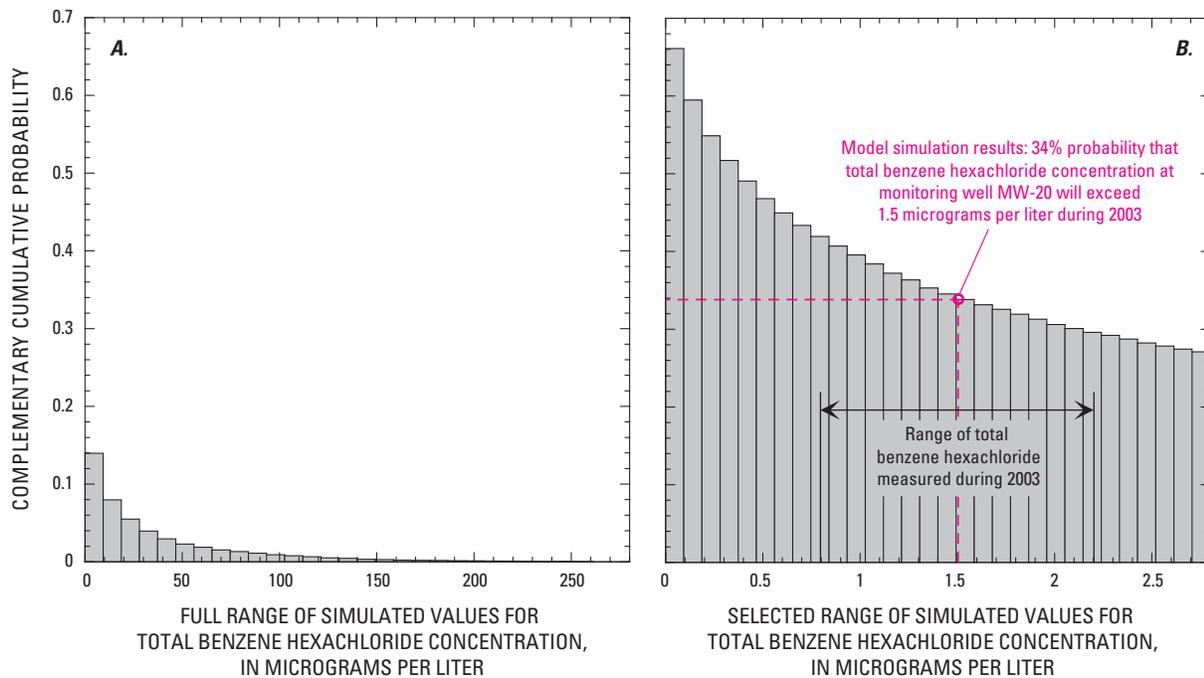
**Figure D5.** Model results from probabilistic analysis at MW-20 during 2003 expressed as a frequency distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values. [MW, monitoring well]



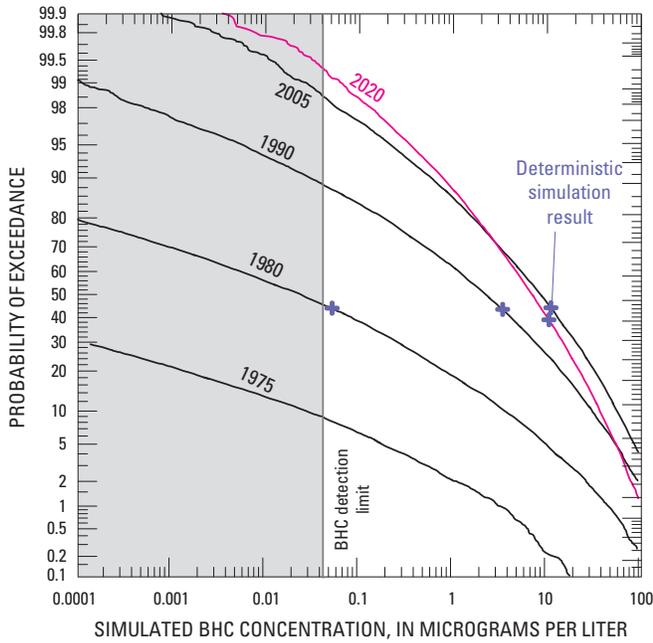
**Figure D6.** Model results from probabilistic analysis at MW-20 during 2003 expressed as a probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values. [MW, monitoring well]



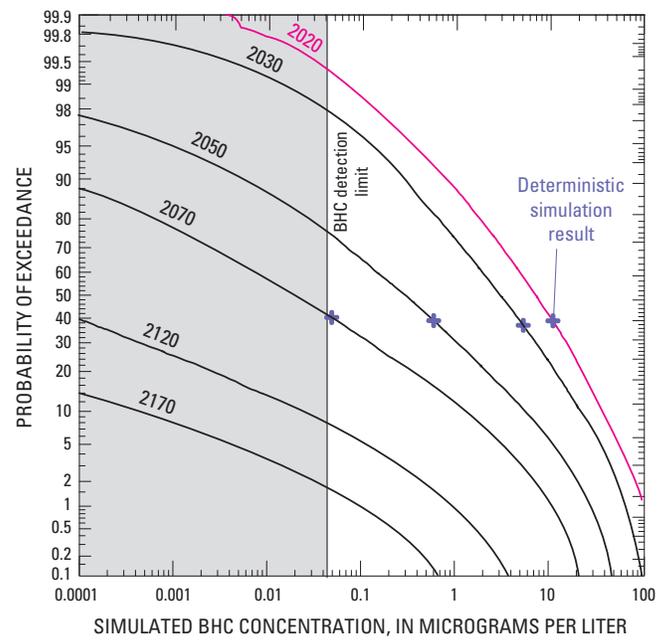
**Figure D7.** Model results from probabilistic analysis at MW-20 during 2003 expressed as a cumulative probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values. [MW, monitoring well]



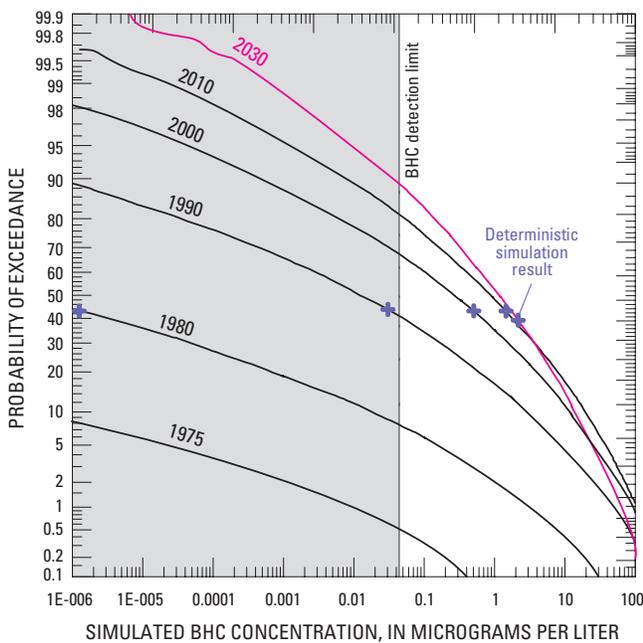
**Figure D8.** Model results from probabilistic analysis at MW-20 during 2003 expressed as a complementary cumulative probability distribution of total benzene hexachloride (BHC) concentration: (A) full range of simulated BHC values and (B) selected range of BHC values. [MW, monitoring well]



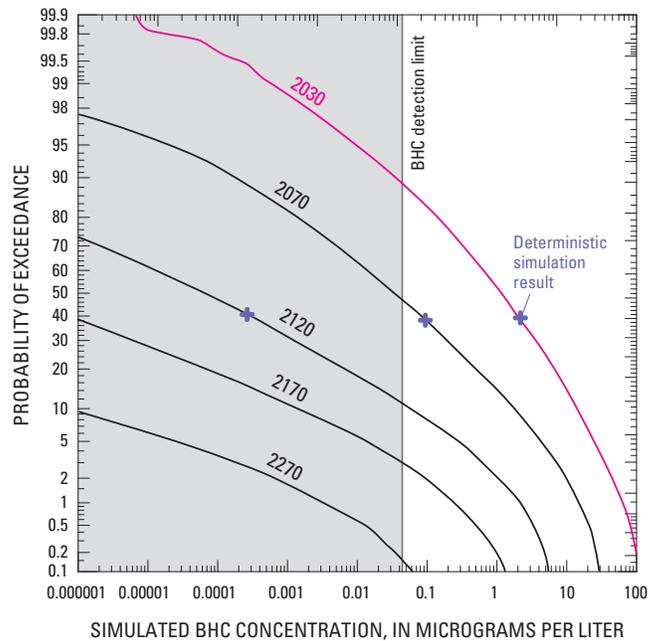
**Figure D9.** Probabilistic model results at MW-17 for time periods before 2020 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



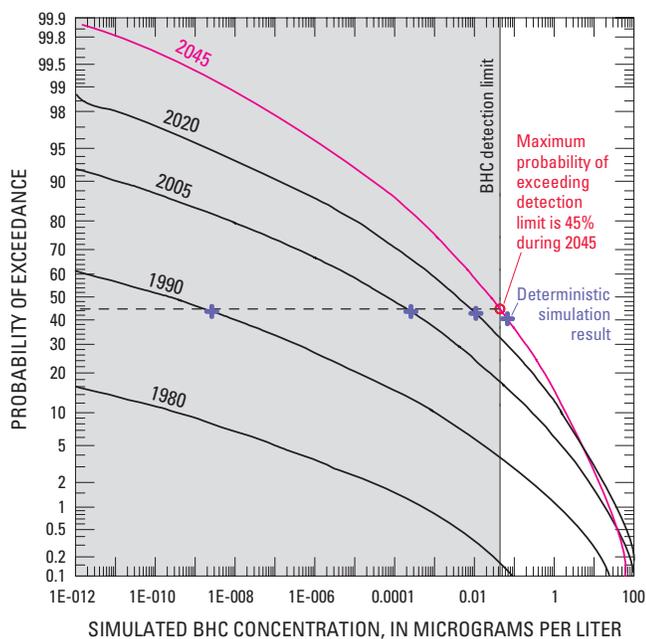
**Figure D10.** Probabilistic model results at MW-17 for time periods after 2020 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



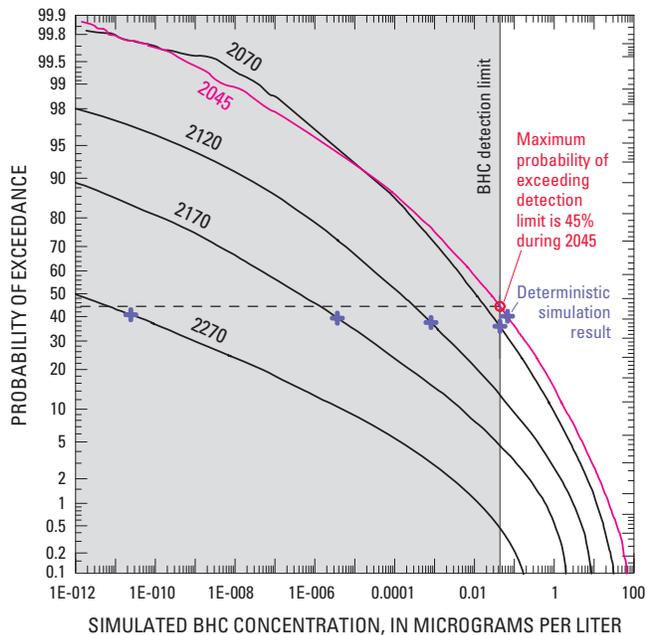
**Figure D11.** Probabilistic model results at MW-20 for time periods before 2030 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



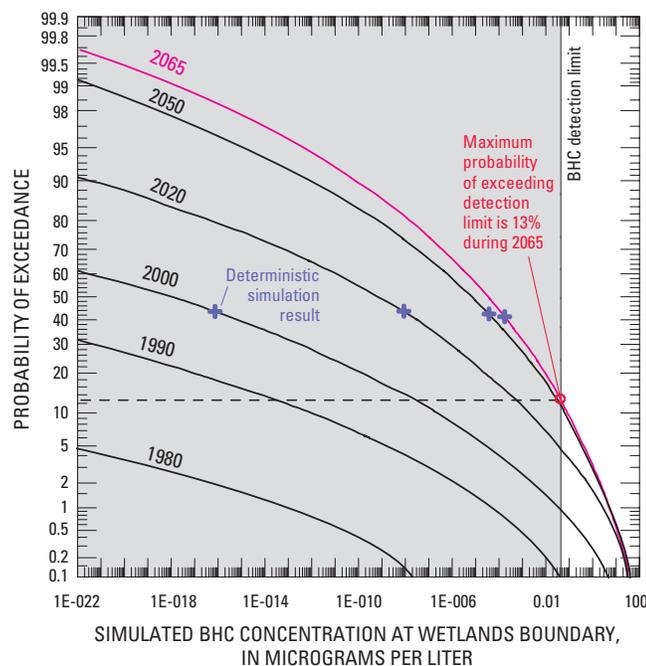
**Figure D12.** Probabilistic model results at MW-20 for time periods after 2030, when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



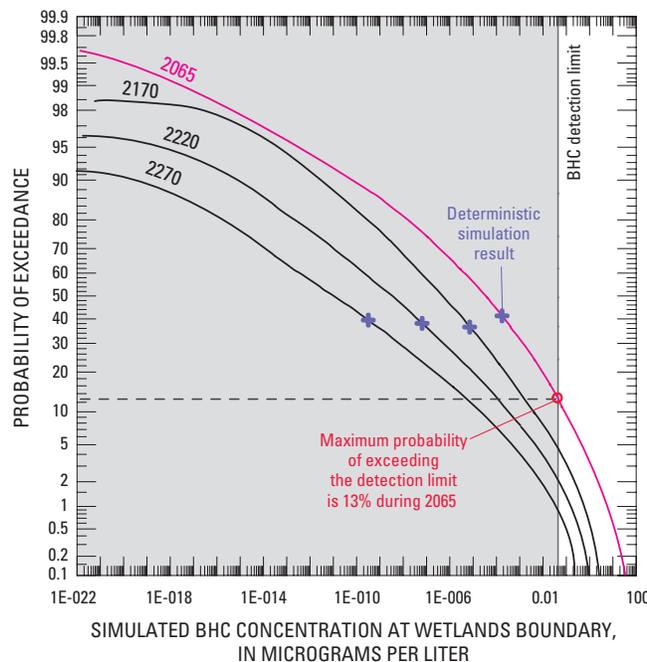
**Figure D13.** Probabilistic model results at MW-23 for time periods before 2045 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



**Figure D14.** Probabilistic model results at MW-23 for time periods after 2045 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



**Figure D15.** Probabilistic model results at the wetlands for time periods before 2065 when the contaminant plume reached the maximum probability of exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



**Figure D16.** Probabilistic model results at the wetlands for time periods after 2065 when the contaminant plume reached exceeding the benzene hexachloride (BHC) detection limit of 0.044 microgram per liter. [MW, monitoring well]



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