Contaminant Candidate List Regulatory Determination Support Document for Aldrin and Dieldrin

Printed on Recycled Paper

Contaminant Candidate List
Regulatory Determination Support Document
for Aldrin and Dieldrin

U.S. Environmental Protection Agency
Office of Water (4607M)
Standards and Risk Management Division
Washington, DC 20460

http://www.epa.gov/SAFEWATER/ccl/cclregdetermine.html

EPA-815-R-03-010
July 2003

Printed on Recycled Paper

Disclaimer

This document is designed to provide supporting information regarding the regulatory determinations for aldrin and dieldrin as part of the Contaminant Candidate List (CCL) evaluation process. This document is not a regulation, and it does not substitute for the Safe Drinking Water Act (SDWA) or the Environmental Protection Agency's (EPA's) regulations. Thus, it cannot impose legally-binding requirements on EPA, States, or the regulated community, and may not apply to a particular situation based upon the circumstances. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

ACKNOWLEDGMENTS

This document was prepared in support of the EPA Office of Ground Water and Drinking Water's regulatory determinations for aldrin and dieldrin as part of the Contaminant Candidate List (CCL) evaluation process. Karen Wirth and Tom Carpenter served as EPA's Co-Team Leaders for the CCL regulatory determination process and Ephraim King as Standards and Risk Management Division Director. Harriet Colbert served as Work Assignment Manager. The CCL Work Group provided technical guidance throughout. In particular, Karen Wirth, Dan Olson, and Joyce Donohue provided scientific and editorial guidance. External expert reviewers and many stakeholders provided valuable advice to improve the CCL Program and this document. The Cadmus Group, Inc., served as the primary contractor providing support for this work. The major contributions of Matt Collins, Emily Brott, Ashton Koo, Richard Zeroka, and Brent Ranalli are gratefully acknowledged. George Hallberg served as Cadmus' Project Manager.

This page intentionally left blank.

USEPA, Office of Water Report: EPA 815-R-03-010, July 2003

CONTAMINANT CANDIDATE LIST REGULATORY DETERMINATION SUPPORT DOCUMENT FOR ALDRIN AND DIELDRIN

EXECUTIVE SUMMARY

Aldrin and dieldrin were 1998 Contaminant Candidate List (CCL) regulatory determination priority contaminants. Aldrin and dieldrin were two of the contaminants considered by the US Environmental Protection Agency (EPA) for a regulatory determination. The available data on occurrence, exposure, and other risk considerations suggest that regulating aldrin and dieldrin with a National Primary Drinking Water Regulation (NPDWR) may not present a meaningful opportunity to reduce health risk. EPA presented preliminary CCL regulatory determinations and further analysis in the June 3, 2002 *Federal Register* Notice (USEPA 2002a; 67 FR 38222) and confirmed the final regulatory determinations in a July 18, 2003 *Federal Register* Notice (USEPA 2003a; 68 FR 42898).

To make this regulatory determination for aldrin and dieldrin, EPA used approaches guided by the National Drinking Water Advisory Council's (NDWAC) Working Group on CCL and Six-Year Review. The Safe Drinking Water Act (SDWA) requirements for National Primary Drinking Water Regulation (NPDWR) promulgation guided protocol development. The SDWA Section 1412(b)(1)(A) specifies that the determination to regulate a contaminant must be based on a finding that each of the following criteria are met: (i) "the contaminant may have adverse effects on the health of persons"; (ii) "the contaminant is known to occur or there is substantial likelihood that the contaminant will occur in public water systems with a frequency and at levels of public health concern"; and (iii) "in the sole judgement of the Administrator, regulation of such contaminant presents a meaningful opportunity for health risk reduction for persons served by public water systems." Available data were evaluated to address each of the three statutory criteria.

Aldrin and dieldrin, related synthetic organic compounds (SOCs), are insecticides that were discontinued for most agricultural uses in 1974 and all uses in 1987. They were used primarily on corn and citrus products, as well as for general crops and timber preservation. In addition, aldrin and dieldrin were used for termite-proofing plywood, building boards, and the plastic and rubber coverings of electrical and telecommunication cables. Aldrin is considered moderately persistent in the environment, with a half-life of approximately 110 days. Dieldrin is among the common degradates of aldrin. Dieldrin is considered extremely persistent, with a half-life of up to seven years.

Aldrin and dieldrin were monitored from 1993 to 1999 under the SDWA Unregulated Contaminant Monitoring (UCM) program. Aldrin and dieldrin are also regulated or monitored by other federal programs including the Clean Water Act Priority Pollutants list, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), and the Resource Conservation and Recovery Act (RCRA). In addition, the Emergency Planning and Community Right-to-Know Act (EPCRA) has listed aldrin as an extremely hazardous substance (EHS) and aldrin is on EPCRA's Toxic Release Inventory (TRI).

Aldrin and dieldrin have been detected at low frequencies and concentrations in ambient surface and ground water and stream bed sediments sampled by the United States Geological Survey's (USGS) National Water Quality Assessment (NAWQA) program. Dieldrin's occurrence is greater in stream bed sediments and biotic tissue. TRI data suggest that aldrin continues to be released to the environment in small quantities even though it is no longer

produced or used commercially in the United States. The presence and persistence of aldrin and dieldrin in the environment is also evidenced by detections at National Priorities List (NPL) hazardous waste sites in at least 31 States and 38 States for aldrin and dieldrin, respectively, and detections in site samples in at least 40 States (both substances) recorded in the Agency for Toxic Substances and Disease Registry's (ATSDR) Hazardous Substance Release and Health Effects Database (HazDat).

Aldrin and dieldrin have also been detected in public water system (PWS) samples collected under SDWA. Occurrence estimates from a cross-section of States with UCM data are very low with only 0.006% and 0.06% of all samples showing detections for aldrin and dieldrin, respectively. Systems with detections of aldrin and dieldrin constitute approximately 0.02% and 0.1% of cross-section systems respectively. Systems with detections above the Health Reference Level (HRL) of 0.002 µg/L, a preliminary health effect level used for this analysis, are also estimated at 0.02% and 0.1% of cross-section systems, because the HRL for aldrin and dieldrin is lower than the Minimum Reporting Level (MRL). National estimates for the population served by PWSs with simple detections (> MRL) and detections above the HRL are also very low, about 40,000 people (roughly 0.02% of the population served by PWSs) for aldrin, and 150,000 people (0.1% of the population) for dieldrin. Using more conservative estimates of occurrence from all States reporting SDWA monitoring data, including States with biased data, 0.2% of the nation's PWSs and 0.5% of the PWS population served (approximately 1,052,000 people) may have aldrin detections > MRL and > HRL, and 0.2% of the nation's PWSs and 0.4% of the PWS population served (approximately 793,000 people) may have comparable detections of dieldrin. Additional SDWA compliance data from the corn belt States of Iowa, Indiana, and Illinois, examined through independent analyses, support the drinking water data analyzed in this report. Surface water detections were low in Indiana and Illinois PWSs, comparable with data obtained through the cross-section model. No detections were reported in the state of Iowa, or in Illinois and Indiana PWS groundwater.

The available toxicological data indicate that aldrin and dieldrin have the potential to cause adverse health effects in humans and animals. In humans, acute aldrin/dieldrin toxicity is most commonly manifested in the central nervous system as hyperirritability, convulsions and coma, in some cases followed by cardiovascular effects such as unusually rapid beating of the heart and elevated blood pressure. In smaller doses over time, aldrin and dieldrin may cause headache, dizziness, general malaise, nausea, vomiting and muscle twitching, or muscle spasms. Aldrin and dieldrin toxicity may also be responsible for reported cases of immunohemolytic and aplastic anemia. Aldrin and dieldrin appear to be weak endocrine disruptors, affecting the reproductive system of animal subjects. Though their carcinogenicity in mice is established, no occupational or epidemiological study has convincingly linked aldrin or dieldrin to cancer in humans.

While there is evidence that aldrin and dieldrin have adverse health effects in humans, their occurrence in drinking water at frequencies or concentrations significant for public health concern is low. Furthermore, occurrence of aldrin and dieldrin in drinking water supplies in the coming years is likely to decrease, since the substances are no longer commercially produced or used. Therefore regulation of aldrin and dieldrin may be unlikely to represent a meaningful opportunity for health risk reduction.

TABLE OF CONTENTS

ACKNOWLEDGMENTS i

EXECUTIVE SUMMARY iii

TABLE OF CONTENTS v

LIST OF TABLES vii

LIST OF FIGURES ix

1.0 INTRODUCTION 1
- 1.1 Purpose and Scope 1
- 1.2 Statutory Framework/Background 1
- 1.3 Statutory History of Aldrin and Dieldrin 2
- 1.4 Regulatory Determination Process 3
- 1.5 Determination Outcome 4

2.0 CONTAMINANT DEFINITION 4
- 2.1 Physical and Chemical Properties 4
- 2.2 Environmental Fate/Behavior 5

3.0 OCCURRENCE AND EXPOSURE 6
- 3.1 Use and Environmental Release 7
  - 3.1.1 Production and Use 7
  - 3.1.2 Environmental Release 7
- 3.2 Ambient Occurrence 8
  - 3.2.1 Data Sources and Methods 8
  - 3.2.2 Results 9
    - 3.2.2.1 Aldrin 9
    - 3.2.2.2 Dieldrin 10
- 3.3 Drinking Water Occurrence 11
  - 3.3.1 Analytical Approach 13
    - 3.3.1.1 UCM Rounds 1 and 2 13
    - 3.3.1.2 Developing a Nationally Representative Perspective 14
      - 3.3.1.2.1 Cross-Section Development 14
      - 3.3.1.2.2 Cross-Section Evaluation 15
    - 3.3.1.3 Data Management and Analysis 16
    - 3.3.1.4 Occurrence Analysis 17
    - 3.3.1.5 Additional Drinking Water Data from the Corn Belt 19
  - 3.3.2 Results 20
    - 3.3.2.1 Aldrin 20
      - 3.3.2.1.1 Occurrence Estimates 20
      - 3.3.2.1.2 Regional Patterns 24
    - 3.3.2.2 Dieldrin 24
      - 3.3.2.2.1 Occurrence Estimates 24
        - 3.3.2.2.2 Regional Patterns 25
- 3.4 Conclusion 25

4.0 HEALTH EFFECTS 26
- 4.1 Hazard Characterization and Mode of Action Implications 26
- 4.2 Dose-Response Characterization and Implications in Risk Assessment 31
- 4.3 Relative Source Contribution 32

4.4 Sensitive Populations 33
4.5 Exposure and Risk Information 33
4.6 Conclusion 34

5.0 TECHNOLOGY ASSESSMENT 34
5.1 Analytical Methods 34
5.2 Treatment Technology 34

6.0 SUMMARY AND CONCLUSIONS - DETERMINATION OUTCOME 36

REFERENCES 39

APPENDIX A: Abbreviations and Acronyms 47

LIST OF TABLES

Table 2-1: Physical and chemical properties 5
Table 3-1: Aldrin detections in stream bed sediments 11
Table 3-2: Dieldrin detections and concentrations in streams and ground water 12
Table 3-3: Dieldrin detections and concentrations in sediments, whole fish, and bivalves (all sites) 12
Table 3-4: Summary occurrence statistics for aldrin 21
Table 3-5: Summary occurrence statistics for dieldrin 27
Table 5-1: Analytical methods for aldrin and dieldrin 35

This page intentionally left blank.

LIST OF FIGURES

Figure 3-1: Geographic distribution of cross-section States for Round 2 (SDWIS/FED) 16
Figure 3-2: States with PWSs with detections of aldrin for all States with data in SDWIS/FED (Round 2) 22
Figure 3-3: Round 2 cross-section States with PWSs with detections of aldrin (any PWSs with results greater than the Minimum Reporting Level [MRL]; above) and concentrations greater than the Health Reference Level (HRL; below) 23
Figure 3-4: States with PWSs with detections of dieldrin for all States with data in SDWIS/FED (Round 2) 28
Figure 3-5: Round 2 cross-section States with PWSs with detections of dieldrin (any PWSs with results greater than the Minimum Reporting Level [MRL]; above) and concentrations greater than the Health Reference Level (HRL; below) 29

This page intentionally left blank.

1.0 INTRODUCTION

1.1 Purpose and Scope

This document presents scientific data and summaries of technical information prepared for, and used in, the Environmental Protection Agency's (EPA) regulatory determinations for aldrin and dieldrin. Information regarding the physical and chemical properties, environmental fate, occurrence and exposure, and health effects of aldrin and dieldrin is included. Analytical methods and treatment technologies are also discussed. Furthermore, the regulatory determination process is described to provide the rationale for the decision.

1.2 Statutory Framework/Background

The Safe Drinking Water Act (SDWA), as amended in 1996, requires the EPA to publish a list of contaminants (referred to as the Contaminant Candidate List, or CCL) to assist in priority-setting efforts. The contaminants included on the CCL were not subject to any current or proposed National Primary Drinking Water Regulations (NPDWR), were known or anticipated to occur in public water systems, and were known or suspected to adversely affect public health. These contaminants therefore may require regulation under SDWA. The first Drinking Water CCL was published on March 2, 1998 (USEPA, 1998a; 63 FR 10274), and a new CCL must be published every five years thereafter.

The 1998 CCL contains 60 contaminants, including 50 chemicals or chemical groups, and 10 microbiological contaminants or microbial groups. The SDWA also requires the Agency to select 5 or more contaminants from the current CCL and determine whether or not to regulate these contaminants with an NPDWR. Regulatory determinations for at least 5 contaminants must be completed 3½ years after each new CCL.

Language in SDWA Section 1412(b)(1)(A) specifies that the determination to regulate a contaminant must be based on a finding that each of the following criteria are met:

Statutory Finding i. the contaminant may have adverse effects on the health of persons;

Statutory Finding ii. the contaminant is known to occur or there is substantial likelihood that the contaminant will occur in public water systems with a frequency and at levels of public health concern; and

Statutory Finding iii. in the sole judgement of the Administrator, regulation of such contaminant presents a meaningful opportunity for health risk reduction for persons served by public water systems.

The geographic distribution of the contaminant is another factor evaluated to determine whether it occurs at the national, regional or local level. This consideration is important because the Agency is charged with developing national regulations and it may not be appropriate to develop NPDWRs for regional or local contamination problems.

EPA must determine if regulating this CCL contaminant will present a meaningful opportunity to reduce health risk based on contaminant occurrence, exposure, and other risk considerations. The Office of Ground Water and Drinking Water (OGWDW) is charged with gathering and analyzing the occurrence, exposure, and risk information necessary to support this regulatory decision. The OGWDW must evaluate when and where this contaminant occurs, and what would be the exposure and risk to public health. EPA must evaluate the impact of potential regulations as well as determine the appropriate measure(s) for protecting public health.

For each of the regulatory determinations, EPA first publishes in the *Federal Register* the draft determinations for public comment. EPA responds to the public comments received, and then finalizes the regulatory determinations. If the Agency finds that regulations are warranted, the regulations must then be formally proposed within 24 months, and promulgated 18 months later. EPA has determined that there is sufficient information to support regulatory determinations for aldrin and dieldrin.

1.3 Statutory History of Aldrin and Dieldrin

Aldrin and dieldrin have been monitored under the SDWA Unregulated Contaminant Monitoring (UCM) program since 1993 (USEPA, 1992; 57 FR 31776). Monitoring ceased for small public water systems (PWSs) under a direct final rule published January 8, 1999 (USEPA, 1999a; 64 FR 1494), and ended for large PWSs with promulgation of the new Unregulated Contaminant Monitoring Regulation (UCMR) issued September 17, 1999 (USEPA, 1999b; 64 FR 50556) and effective January 1, 2001. At the time the UCMR lists were developed, the Agency concluded there were adequate monitoring data for regulatory determinations. This obviated the need for continued monitoring under the new UCMR list.

EPA previously recommended guidelines for exposure to aldrin and dieldrin in drinking water through health advisories issued in 1991 and 1988, respectively (USEPA, 1991a; USEPA, 1988). As part of the CCL process, health effects data have been reviewed. These are summarized in section 4.0 of this document.

Aldrin and dieldrin are regulated or monitored by other federal programs as well. They are included on the Clean Water Act Priority Pollutants list for which the EPA establishes ambient water quality criteria. The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or "Superfund") includes aldrin and dieldrin as hazardous substances while the Resource Conservation and Recovery Act (RCRA) classifies them as hazardous constituents (USEPA, 2000e). CERCLA's listing requires reporting of releases over a certain "reportable quantity" which, for aldrin and dieldrin, is one pound (USEPA, 1996).

In addition, the Emergency Planning and Community Right-to-Know Act (EPCRA) has listed aldrin as an extremely hazardous substance (EHS). The presence of EHSs in excess of the Threshold Planning Quantity (TPQ), requires certain emergency planning activities to be conducted. For aldrin, the Threshold Planning Quantity is 500 lbs if it is in molten form, in solution, or in powder form with particle size less than 100 microns. Otherwise, aldrin's TPQ is 10,000 lbs (USEPA, 1996). Aldrin is also on EPCRA's Toxic Release Inventory (TRI). The TRI requires certain industrial sectors to publicly report the environmental release or transfer of listed chemicals (USEPA, 2000c).

The Occupational Safety and Health Administration (OSHA) recommends occupational exposure limits for aldrin and dieldrin of 250 $\mu g/m^3$ for an 8-hour workday over a 40-hour workweek. The National Institute for Occupational Safety and Health (NIOSH) suggests that aldrin and dieldrin levels be kept at the lowest amounts that can be measured reliably (ATSDR, 1993).

The U.S. Department of Agriculture (USDA) discontinued registration and use of aldrin and dieldrin as pesticides in 1970. In 1972, EPA canceled all but the following three uses of aldrin and dieldrin: subsurface ground insertion for termite control, the dipping of nonfood plant roots and tops, and moth-proofing in manufacturing processes using completely closed systems. This cancellation decision was finalized in 1974. The sole manufacturer of aldrin and dieldrin, Shell Chemical Company, ceased producing the compounds for moth-proofing and the dipping of non-food plants in 1974, and canceled their use as termiticides in 1987 (ATSDR, 1993).

1.4 Regulatory Determination Process

In developing a process for the regulatory determinations, EPA sought input from experts and stakeholders. EPA asked the National Research Council (NRC) for assistance in developing a scientifically sound approach for deciding whether or not to regulate contaminants on the current and future CCLs. The NRC's Committee on Drinking Water Contaminants recommended that EPA: (1) gather and analyze health effects, exposure, treatment, and analytical methods data for each contaminant; (2) conduct a preliminary risk assessment for each contaminant based on the available data; and (3) issue a decision document for each contaminant describing the outcome of the preliminary risk assessment. The NRC noted that in using this decision framework, EPA should keep in mind the importance of involving all interested parties.

One of the formal means by which EPA works with its stakeholders is through the National Drinking Water Advisory Council (NDWAC). The NDWAC comprises members of the general public, State and local agencies, and private groups concerned with safe drinking water, and advises the EPA Administrator on key aspects of the Agency's drinking water program. The NDWAC provided specific recommendations to EPA on a protocol to assist the Agency in making regulatory determinations for current and future CCL contaminants. Separate but similar protocols were developed for chemical and microbial contaminants. These protocols are intended to provide a consistent approach to evaluating contaminants for regulatory determination, and to be a tool that will organize information in a manner that will communicate the rationale for each determination to stakeholders. The possible outcomes of the regulatory determination process are: a decision to regulate, a decision not to regulate, or a decision that some other action is needed (e.g., issuance of guidance).

The NDWAC protocol uses the three statutory requirements of SDWA Section 1412(b)(1)(A)(i)-(iii) (specified in section 1.2) as the foundation for guiding EPA in making regulatory determination decisions. For each statutory requirement, evaluation criteria were developed and are summarized below.

To address whether a contaminant may have adverse effects on the health of persons (statutory requirement (i)), the NDWAC recommended that EPA characterize the health risk and estimate a health reference level for evaluating the occurrence data for each contaminant.

Regarding whether a contaminant is known to occur, or whether there is substantial likelihood that the contaminant will occur, in public water systems with a frequency, and at levels, of public health concern (statutory requirement (ii)), the NDWAC recommended that EPA consider: (1) the actual and estimated national percent of public water systems (PWSs) reporting detections above half the health reference level; (2) the actual and estimated national percent of PWSs with detections above the health reference level; and (3) the geographic distribution of the contaminant.

To address whether regulation of a contaminant presents a meaningful opportunity for health risk reduction for persons served by public water systems (statutory requirement (iii)) the NDWAC recommended that EPA consider estimating the national population exposed above half the health reference level and the national population exposed above the health reference level.

The approach EPA used to make regulatory determinations followed the general format recommended by the NRC and the NDWAC to satisfy the three SDWA requirements under Section 1412(b)(1)(A)(i)-(iii). The process was independent of many of the more detailed and comprehensive risk management factors that will influence the ultimate regulatory decision making process. Thus, a decision to regulate is the beginning of the Agency regulatory development process, not the end.

Specifically, EPA characterized the human health effects that may result from exposure to a contaminant found in drinking water. Based on this characterization, the Agency estimated a health reference level (HRL) for each contaminant.

For each contaminant EPA estimated the number of PWSs with detections >½HRL and >HRL, the population served at these benchmark values, and the geographic distribution, using a large number of occurrence data (approximately seven million analytical points) that broadly reflect national coverage. Round 1 and Round 2 UCM data, evaluated for quality, completeness, bias, and representativeness, were the primary data used to develop national occurrence estimates. Use and environmental release information, additional drinking water data sets (e.g., State drinking water data sets, EPA National Pesticide Survey, and Environmental Working Group data reviews), and ambient water quality data (e.g., NAWQA, State and regional studies, and the EPA Pesticides in Ground Water Database) were also consulted.

The findings from these evaluations were used to determine if there was adequate information to evaluate the three SDWA statutory requirements and to make a determination of whether to regulate a contaminant.

1.5 Determination Outcome

After reviewing the best available public health and occurrence information, EPA has made a determination not to regulate the contaminants aldrin and dieldrin with NPDWRs. EPA has found that aldrin and dieldrin may have adverse effects on the health of persons and are known to occur in at least some PWSs. However, EPA has not found that they occur in public water systems at frequencies or levels of public health concern. Furthermore, all uses of these compounds were suspended in 1987. EPA does not consider exposure to aldrin or dieldrin to be widespread nationally. All CCL regulatory determinations are formally presented in the *Federal Register* Notices (USEPA, 2002a; 67 FR 38222; and USEPA, 2003a; 68 FR 42898). The following sections summarize the data used by the Agency to reach this decision.

2.0 CONTAMINANT DEFINITION

Aldrin and dieldrin are both synthetic organic compounds (SOCs) that are white powders when pure, but tan powders in their technical-grade forms. Aldrin is generated by the Diels-Alder condensation of hexachlorocyclopentadiene with bicyclo[2.2.1]-2,5-heptadiene. Its chemical name is 1,2,3,4,10,10-hexachloro-1,4,4α,5,8,8α-hexahydro-1,4-endo,exo-5,8-dimethanonaphthalene (abbreviated HHDN). Dieldrin is generated by the epoxidation of aldrin. The chemical name for dieldrin is 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4α,5,6,7,8,8α-octahydro-1,4-endo,exo-5,8-dimethanonaphthalene (abbreviated HEOD). The Shell Chemical Company was the sole United States manufacturer and distributor of these compounds. Trade names for aldrin include: Aldrec, Aldrex, Drinox, Octalene, Seedrin, and Compound 118. Some dieldrin trade names are: Alvit, Dieldrix, Octalox, Quintox, and Red Shield (ATSDR, 1993). Aldrin and dieldrin are insecticides that were discontinued for all uses in 1987. Aldrin and dieldrin combat insects by contact or ingestion, and were used primarily on corn and citrus products, as well as for general crops and timber preservation. In addition, aldrin and dieldrin were used for termite-proofing plywood, building boards, and the plastic and rubber coverings of electrical and telecommunication cables (ATSDR, 1993).

2.1 Physical and Chemical Properties

Table 2-1 lists summary information regarding the physical and chemical properties of aldrin and dieldrin. Also included are their CAS Registry Numbers and molecular formulas.

Table 2-1: Physical and chemical properties

Identification	**Aldrin**	**Dieldrin**
CAS number	309-00-2	60-57-1
Molecular Formula	$C_{12}H_8Cl_6$	$C_{12}H_8Cl_6O$
Physical and Chemical Properties	**Aldrin**	**Dieldrin**
Boiling Point	145 °C	330 °C
Melting Point	104 °C pure, 40-60 °C technical grade	175-176 °C pure, 95 °C technical grade
Molecular Weight	364.93 g/mol	380.93 g/mol
Log K_{oc}	4.69	3.87
Log K_{ow}	6.5*	4.55
Water Solubility	0.20 mg/L at 25 °C	0.18 mg/L at 25 °C
Vapor Pressure	1.4×10^{-4} mm Hg at 25 °C	7.78×10^{-7} mm Hg at 25 °C
Henry's Law Constant †	1.30×10^{-2}	6.16×10^{-4}

*after ATSDR, 1993; *Howard, 1991.*
†note: this quantity is expressed in a dimensionless form.

2.2 Environmental Fate/Behavior

Aldrin readily converts to dieldrin in aerobic terrestrial and aqueous environments, as well as in the atmosphere. Aldrin has a low leaching potential (or high soil/water partitioning) and does not volatilize appreciably from subsurface soil particles (50% loss after 10-15 weeks). Its volatilization is more rapid at the soil surface (50% loss after 1-2 weeks), causing significant loss to the atmosphere post-application. Dieldrin also does not volatilize readily from soil, although volatilization is still an important loss process for this persistent compound. Dieldrin sorbs strongly to soils and resists leaching to ground water systems, but can be present in run-off to surface water, sorbed to waterborne sediments (ATSDR, 1993; Howard, 1991). Laboratory studies suggest that aldrin rapidly degrades in the atmosphere to photoaldrin, dieldrin, or photodieldrin (ATSDR, 1993). Dieldrin is broken down by hydroxyl radicals in the atmosphere with an estimated half life of one day. Dieldrin is more stable to this degradation when attached to particulate matter in the atmosphere, and can thus be transported long distances before it is lost through wet or dry deposition. Due to this transport mechanism and to the compound's resistance to abiotic degradation and biotransformation, dieldrin is persistent and widespread in the environment (ATSDR, 1993; Howard, 1991).

Aldrin is classified as a moderately persistent compound, which signifies that its half life in soil ranges from 20-100 days (Howard, 1991). Aldrin degrades to dieldrin by epoxidation in all aerobic and biologically active soils. Another metabolite of aldrin in soil is aldrin acid (ATSDR, 1993). Mathematical models estimate that aldrin would degrade by 69% to dieldrin in 81 days

when applied within 15 cm from the soil surface. The half life of aldrin was estimated to be approximately 110 days when applied to soil at a rate of 1.1-3.4 kg/hectare, with 95% disappearance in 3 years. The half life of aldrin was shown to be significantly lower in sandy loam compared to clay loam soils (36.5 versus 97 days when applied at a rate of 9 kg/hectare; ATSDR, 1993).

Dieldrin is extremely persistent in soil, with a residence time of often greater than 7 years. Although volatilization is dieldrin's most important terrestrial loss process, its low vapor pressure keeps it from evaporating quickly from soil. Another loss process from soil is adsorption on dust particles and atmospheric transport (Howard, 1991). An experiment with dieldrin applied to a microagroecosystem demonstrated that 73% of the compound remained on the soil surface after 11 days. Soil moisture content also affects dieldrin volatilization rates. One study showed that after five months, about 18% of dieldrin had volatilized from moist land compared to 2% from flooded and 7% from dry lands (ATSDR, 1993). Volatilization rates for dieldrin decrease with time and increase with increasing temperature to a maximum at 25°C. Dieldrin is highly resistant to biodegradation in soil, with an estimated half life in soil of 868 days (estimated using the Henry's law constant and the organic carbon partition coefficient, K_{oc}; ATSDR, 1993).

Aldrin sorbs to soil and is not available to leach to ground water systems, as evidenced by its general absence in ground water samples (ATSDR, 1993; Howard, 1991). However, small amounts of aldrin have been detected in surface waters. Volatilization is expected to be a significant loss process for aldrin in water, with the rate of volatilization directly proportional to wind speed and current velocity and inversely proportional to water body depth (Howard, 1991). Aldrin was photooxidized by 75% to dieldrin following 48 hours of irradiation at 238 nm in filtered natural field water. Studies show that aldrin degrades under anaerobic conditions in biologically active wastewater sludge (pH 7-8, 35° C) with a half life of under one week (ATSDR, 1993).

Dieldrin is also mostly absent from ground water samples because of its tendency to sorb to soil. Dieldrin that is sorbed to waterborne sediments can run-off to surface waters. Dieldrin does not undergo hydrolysis or significant aqueous biodegradation. Conflicting data exist regarding the importance of volatilization for dieldrin in water (Howard, 1991; ATSDR, 1993). Dieldrin is slowly converted in the presence of sunlight to photodieldrin, a stereoisomer of the compound (water half-life of 4 months; Howard, 1991). Dieldrin added to natural water (from a drainage canal in an agricultural area) and incubated in the dark degraded by less than 20% in 16 weeks. Dieldrin is stable against significant degradation in both biologically active and anaerobic wastewater sludge (89% remained after 48 hours of continuous anaerobic digestion; 45% remained after 9 days of aerobic digestion; ATSDR, 1993).

Aldrin and dieldrin have ranges of log K_{ow} values from 5.68 to 7.4, and from 4.32 to 6.2, respectively, suggesting high potentials for bioaccumulation. Dieldrin's extreme nonpolarity gives it a strong affinity towards animal fat and plant waxes (ATSDR, 1993; Howard ,1991).

3.0 OCCURRENCE AND EXPOSURE

This section examines the occurrence of aldrin and dieldrin in drinking water. While no complete national database exists of unregulated or regulated contaminants in drinking water from public water systems (PWSs) collected under the Safe Drinking Water Act (SDWA), this report aggregates and analyzes existing State data that have been screened for quality, completeness, and representativeness. Populations served by PWSs exposed to aldrin and dieldrin are estimated, and the occurrence data are examined for regional or other special trends. To augment the incomplete national drinking water data and aid in the evaluation of occurrence,

information on the use and environmental release, as well as ambient occurrence of aldrin and dieldrin, is also reviewed.

3.1 Use and Environmental Release

3.1.1 Production and Use

Aldrin and dieldrin combat insects by contact or ingestion, and were used primarily on corn and citrus products, as well as for general crops and timber preservation. In addition, aldrin and dieldrin were used for termite-proofing plywood, building boards, and the plastic and rubber coverings of electrical and telecommunication cables (ATSDR, 1993). In 1972, EPA canceled all but the following three uses of aldrin and dieldrin: subsurface ground insertion for termite control, the dipping of non-food plant roots and tops, and moth-proofing in manufacturing processes using completely closed systems. This cancellation decision was finalized in 1974.

Aldrin was not imported into the United States prior to the 1974 cancellation decision, however Shell International (Holland) imported the chemical for limited use from 1974 to 1985 (with the exception of 1979 and 1980, when imports were temporarily suspended). An estimated 1-1.5 million lbs of aldrin were imported annually from 1981-1985, after which time importation ceased. No importation data were found for dieldrin. By 1987, all uses of aldrin and dieldrin had been canceled voluntarily by the manufacturer (ATSDR, 1993).

3.1.2 Environmental Release

Aldrin is listed as a Toxic Release Inventory chemical. In 1986, the Emergency Planning and Community Right-to-Know Act (EPCRA) established the Toxic Release Inventory (TRI) of hazardous chemicals. Created under the Superfund Amendments and Reauthorization Act (SARA) of 1986, EPCRA is also sometimes known as SARA Title III. The EPCRA mandates that larger facilities publicly report when TRI chemicals are released into the environment. This public reporting is required for facilities with more than 10 full-time employees that annually manufacture or produce more than 25,000 pounds, or use more than 10,000 pounds, of a TRI chemical (USEPA, 1996; USEPA, 2000c).

Under these conditions, facilities are required to report the pounds per year of aldrin released into the environment both on- and off-site. The production, import, and use of aldrin had been canceled by the time the TRI was instated, however, and therefore no release or transfer data were reported. In 1995, Resource Conservation and Recovery Act (RCRA) subtitle C hazardous waste treatment and disposal facilities were added to the list of those facilities required to present release data to the TRI. This addition became effective for the 1998 reporting year, which is the most recent TRI data currently available. Waste treatment facilities from three States (AR, MI, TX) reported releases of aldrin in 1998, with on- and off-site releases totaling 25,622 pounds. The on-site quantity is subdivided into air emissions, surface water discharges, underground injections, and releases to land. Most of the aldrin released to the environment was released directly to land (22,000 lbs; USEPA, 2000a).

Although the TRI data can be useful in giving a general idea of release trends, it is far from exhaustive and has significant limitations. For example, only industries that meet TRI criteria (at least 10 full-time employees and the manufacture and processing of quantities exceeding 25,000 lbs/yr, or use of more than 10,000 lbs/yr) are required to report releases. These reporting criteria do not account for releases from smaller industries. Also, the TRI data is meant to reflect releases and should not be used to estimate general exposure to a chemical (USEPA, 2000b).

Aldrin and dieldrin are included in the Agency for Toxic Substances and Disease Registry's (ATSDR) Hazardous Substance Release and Health Effects Database (HazDat). This database

records detections of listed chemicals in site samples; aldrin and dieldrin were both detected in 40 States. The National Priorities List (NPL) of hazardous waste sites, created in 1980 by the Comprehensive Environmental Response, Compensation & Liability Act (CERCLA), is a listing of some of the most health-threatening waste sites in the United States. Aldrin and dieldrin were detected in NPL hazardous waste sites in 31 and 38 States, respectively (USEPA, 1999c).

In summary, aldrin and dieldrin have not been produced in the United States since 1974, and all uses of the pesticides were canceled by 1987. The chemicals had been used mostly on corn and citrus products. Aldrin was imported to the United States from Holland from 1974-1985 (with the exception of 1979 and 1980) in quantities of approximately 1-1.5 million lbs/yr. TRI data from 1998 suggest that aldrin continues to be released into the environment, even though the chemical is no longer produced nor used in the United States. The presence and persistence of aldrin and dieldrin in the environment is evidenced by detections of the compounds in hazardous waste sites in at least 31 and 38 States, respectively (at NPL sites), as well as detections of both chemicals in site samples in at least 40 States (listed in ATSDR's HazDat).

3.2 Ambient Occurrence

To understand the presence of a chemical in the environment, an examination of ambient occurrence is useful. In a drinking water context, ambient water is source water existing in surface waters and aquifers before treatment. The most comprehensive and nationally consistent data describing ambient water quality in the United States are being produced through the United States Geological Survey's (USGS) National Water Quality Assessment (NAWQA) program. (NAWQA, however, is a relatively young program and complete national data are not yet available from their entire array of sites across the nation.)

3.2.1 Data Sources and Methods

The USGS instituted the NAWQA program in 1991 to examine water quality status and trends in the United States. NAWQA is designed and implemented in such a manner as to allow consistency and comparison between representative study basins located around the country, facilitating interpretation of natural and anthropogenic factors affecting water quality (Leahy and Thompson, 1994).

The NAWQA program consists of 59 significant watersheds and aquifers referred to as "study units." The study units represent approximately two thirds of the overall water usage in the United States and a similar proportion of the population served by public water systems. Approximately one half of the nation's land area is represented (Leahy and Thompson, 1994).

To facilitate management and make the program cost-effective, approximately one third of the study units at a time engage in intensive assessment for a period of 3 to 5 years. This is followed by a period of less intensive research and monitoring that lasts between 5 and 7 years. This way all 59 study units rotate through intensive assessment over a ten-year period (Leahy and Thompson, 1994). The first round of intensive monitoring (1991-96) targeted 20 units and the second round monitored another 16 beginning in 1994. This first group was more heavily slanted toward agricultural basins. A national synthesis of results from these study units, focusing on pesticides and nutrients, has been compiled and analyzed (Kolpin et al., 2000; Larson et al., 1999; USGS, 1999a).

Aldrin was not an analyte for either the ground water or the surface water NAWQA studies included in the pesticide and nutrient national synthesis (Kolpin et al., 1998; Larson et al., 1999; USGS, 1999b). Because of analytical and budget constraints the NAWQA program targets certain pesticides, many of which are high use and/or have potential environmental significance (Larson et al., 1999; USGS, 1999a). Aldrin may have been excluded because it breaks down in

the environment to dieldrin (among other degradates), which is analyzed in the NAWQA studies (USGS, 1999b). Aldrin persisting in the environment is more likely to be found in sediments or biotic tissues because of its strong hydrophobicity and sorption potential (ATSDR, 1993; Nowell, 1999; USGS, 2000). Consequently, NAWQA investigators focused their aldrin occurrence studies on bed sediments and aquatic biota tissue (Nowell, 1999).

Dieldrin is an analyte for both surface and ground water NAWQA studies. Two of the first 20 study basins analyzed in the pesticide and nutrient national synthesis reports, the Central Nebraska Basins and the White River Basin in Indiana, are located in the corn belt where dieldrin use was heavy in the 1960s. The Minimum Reporting Level (MRL) for dieldrin is 0.001 µg/L (Kolpin et al., 1998), substantially lower than most drinking water monitoring.

Data are also available for dieldrin occurrence in surface water in the Mississippi River and six major tributaries draining corn belt States (Goolsby and Battaglin, 1993). These data are the result of a USGS regional water quality investigation, and details regarding sampling and analytical methods are described in the report.

Aldrin and dieldrin are organochlorine insecticides. As a group, organochlorines are hydrophobic and resist degradation. Hydrophobic ("water hating") compounds have low water solubilities and strong tendencies to sorb to organic material in sediments and accumulate in the tissue of aquatic biota, where they can persist for long periods of time (ATSDR, 1993; USGS, 2000). Organochlorines may be present in bed sediments and tissues of aquatic systems even when they are undetectable in the water column using conventional methods (Nowell, 1999). The occurrence of a toxic compound in stream sediments is pertinent to drinking water concerns because some desorption of the compound from sediments into water will occur through equilibrium reactions, although in very low concentrations.

To determine their presence in hydrologic systems of the United States, the NAWQA program has investigated organochlorine pesticide detections in bed sediments and biotic tissue, focusing on the organochlorine insecticides that were used heavily in the past (Nowell, 1999). The occurrence of aldrin, one of the top three insecticides used for agriculture in the 1960s and widely used to kill termites in structures until the mid 1980s, and dieldrin was investigated in this study (Nowell, 1999; USGS, 1999a). Sampling was conducted at 591 sites from 1992-1995 in the 20 NAWQA study units where the first round of intensive assessment took place. Two of these basins, the Central Nebraska Basins and the White River Basin in Indiana, are located in the corn belt where aldrin use was heavy in the 1960s. Details regarding sampling techniques and analytical methods are described by Nowell (1999).

3.2.2 Results

3.2.2.1 Aldrin

Aldrin was not detected in aquatic biota tissue samples. However, it was detected in stream bed sediment samples. The occurrence frequencies above the Minimum Reporting Level (MRL) of 1 µg/kg, and basic summary statistics, indicate that occurrence in sediments is very low (Table 3-1). Both the median and 95th percentile concentrations were reported as non-detections (< MRL) across all land use categories.

Aldrin was detected in sediments only at agricultural or mixed land use sites, perhaps reflecting the heavy agricultural use in the late 1960s and early 1970s (Table 3-1). Interesting in light of the more recent termiticide use, no urban detections were reported. This may be partly a function of the NAWQA sampling design that targeted basins more representative of agricultural and mixed land use conditions for the first round of intensive monitoring from which these

sediment data were produced (see section 3.2.1 above) (Data from later rounds are not yet available)

3.2.2.2 Dieldrin

Detection frequencies and concentrations of dieldrin in ambient surface and ground water are low, especially in ground water which is the case for insecticides in general (Table 3-2; Kolpin et al., 1998; Miller and Wilber, 1999). However, using a common reporting limit of 0.01 µg/L, dieldrin is the most commonly detected insecticide in ground water in these USGS studies. This possibly reflects the historically heavy use of aldrin and dieldrin and clearly indicates dieldrin's environmental persistence (Kolpin et al., 1998; Miller, 2000). Also, though relatively immobile in water when compared to newer pesticides, dieldrin is one of the most mobile of the older organochlorine pesticides (USGS, 1999a).

Dieldrin detection frequencies are considerably higher in shallow ground water in urban areas as compared to shallow ground water in agricultural areas (Table 3-2), a likely consequence of the more recent use of aldrin and dieldrin as a termiticide and industrial moth-proofing agent until the mid-1980s. Agricultural uses were discontinued in the 1970s. Major aquifers, which are generally deep, have very low detection frequencies and concentrations of dieldrin. Hydrophobic compounds have high sorption potential and are not very mobile in ground water, making their occurrence in deep aquifers unlikely.

In streams, detection frequencies are higher compared to ground water (Table 3-2). Dieldrin's chemical characteristics, chiefly its hydrophobicity, make it less likely to be transported to the subsurface with ground water recharge. Instead, dieldrin sorbs easily to sediments and biotic tissues and may persist in surface water environments for many years after applications have ceased. Differences in detection frequencies and concentrations between urban and agricultural settings are less pronounced for streams than for ground water, but frequencies and concentrations are greater for streams in agricultural settings.

The concentrations and detection frequencies of dieldrin in bed sediments and biotic tissues are considerably higher than water, although the median concentration of all samples is still below the MRL (Table 3-3). Occurrence of dieldrin is highest in whole fish, highlighting the potential for it to bioaccumulate (Kolpin et al., 1998). The trend of higher concentrations and detection frequencies in urban environments is again apparent when examining dieldrin occurrence across various land use settings in sediments and biotic tissues. Urban areas have the highest detections and concentrations. Occurrence in agricultural and mixed land use types is lower and approximately equivalent. Forest and rangeland show very low occurrence (Nowell, 1999).

While concentrations in water are generally low, a risk-specific dose (RSD) criteria of 0.02 µg/L, a concentration associated with a cancer risk level of 1 in 100,000 people, was exceeded at one site at least, in both surface and ground water (Kolpin et al., 1998; Larson et al., 1999; USGS, 1998).

A USGS regional water quality investigation provides additional information on the occurrence of dieldrin in the corn belt. For surface water sampling from April 1991 to March 1992 from the Mississippi River and six tributaries draining the corn belt, 8% of all samples and 71% of sites had detections greater than the reporting limit of 0.02 µg/L. The maximum concentration was approximately 0.03 µg/L (Goolsby and Battaglin, 1993).

Table 3-1: Aldrin detections in stream bed sediments

	Detection frequency (% samples > MRL of 1 µg/kg)	Concentration (all samples; µg/kg dry weight)		
		median	95th percentile	maximum
urban	0.0 %	nd*	nd	nd
mixed	0.5 %	nd	nd	3
agricultural	0.6 %	nd	nd	2.2
forest-rangeland	0.0 %	nd	nd	nd
all sites	0.4 %	nd	nd	3

after Nowell, 1999
**not detected in concentration greater than MRL*

3.3 Drinking Water Occurrence

The SDWA, as amended in 1986, required public water systems (PWSs) to monitor for specified "unregulated" contaminants, on a five year cycle, and to report the monitoring results to the States. Unregulated contaminants do not have an established or proposed NPDWR, but they are contaminants that were formally listed and required for monitoring under federal regulations. The intent was to gather scientific information on the occurrence of these contaminants to enable a decision as to whether or not regulations were needed. All non-purchased community water systems (CWSs) and non-purchased non-transient non-community water systems (NTNCWSs), with greater than 150 service connections, were required to conduct this unregulated contaminant monitoring. Smaller systems were not required to conduct this monitoring under federal regulations, but were required to be available to monitor if the State decided such monitoring was necessary. Many States collected data from smaller systems. Additional contaminants were added to the Unregulated Contaminant Monitoring (UCM) program in 1991 (USEPA, 1991b; 56 FR 3526) for required monitoring that began in 1993 (USEPA, 1992; 57 FR 31776).

Table 3-2: Dieldrin detections and concentrations in streams and ground water

	Detection frequency (% samples ≥ MRL*)		Concentration percentiles (all samples; µg/L)		
	% ≥ 0.001 µg/L	% ≥ 0.01 µg/L	median	95th	maximum
streams					
urban	3.67 %	1.83 %	nd**	nd	0.016
integrator	3.27 %	1.63 %	nd	nd	0.015
agricultural	6.90 %	3.90 %	nd	0.007	0.027
all sites	4.64 %	2.39 %	nd	nd	0.19
ground water					
shallow urban	5.65 %	3.32 %	nd	0.005	0.068
shallow agricultural	0.97 %	0.65 %	nd	nd	0.057
major aquifers	0.43 %	0.21 %	nd	nd	0.03
all sites	1.42 %	0.93 %	nd	nd	0.068

after USGS, 1998
**MRL for dieldrin in water studies:0.001 µg/L.*
***not detected in concentrations greater than MRL*

Table 3-3: Dieldrin detections and concentrations in sediments, whole fish, and bivalves (all sites)

	Detection frequency (% samples > MRL*)	Concentration percentiles (all samples; µg/kg dry weight)		
		median	95th	maximum
sediments	13.7 %	nd**	2.7	18
whole fish	28.6 %	nd	31.9	260
bivalves	6.4 %	nd	6.4	20

after Nowell, 1999
** MRL for dieldrin in sediments: 1 µg/kg; dieldrin in whole fish and bivalves: 5 µg/kg.*
***not detected in concentrations greater than MRL*

Aldrin and dieldrin have been monitored under the SDWA Unregulated Contaminant Monitoring (UCM) program since 1993 (USEPA, 1992; 57 FR 31776). Monitoring ceased for small public water systems (PWSs) under a direct final rule published January 8, 1999 (USEPA, 1999a; 64 FR 1494), and ended for large PWSs with promulgation of the new Unregulated Contaminant Monitoring Regulation (UCMR) issued September 17, 1999 (USEPA, 1999b; 64 FR 50556) and effective January 1, 2001. At the time the UCMR lists were developed, the Agency concluded there were adequate monitoring data for a regulatory determination. This obviated the need for continued monitoring under the new UCMR list.

3.3.1 Analytical Approach

Currently, there is no complete national record of unregulated or regulated contaminants in drinking water from PWSs collected under SDWA. Many States have submitted unregulated contaminant PWS monitoring data to EPA databases, but there are issues of data quality, completeness, and representativeness. Nonetheless, a significant amount of State data are available for UCM contaminants that can provide estimates of national occurrence. The contaminant occurrence analyses findings presented in this report are based on a national cross-section of aggregated State data (i.e., a representative subset of available State data) derived from the Federal Safe Drinking Water Information System (SDWIS/FED) database.

The National Contaminant Occurrence Database (NCOD) is an interface to the actual occurrence data stored in the Safe Drinking Water Information System (Federal version; SDWIS/FED) and can be queried to provide a summary of the data in SDWIS/FED for a particular contaminant. The drinking water occurrence data for aldrin and dieldrin presented here were derived from monitoring data available in the SDWIS/FED database. Note, however, that the SDWIS/FED data used in this report have undergone significant review, edit, and filtering to meet various data quality objectives for the purposes of this analysis. Hence, not all data from a particular source were used; only data meeting the quality objectives described below were included. The sources of these data, their quality and national aggregation, and the analytical methods used to estimate a given contaminant's national occurrence (from these data) are discussed in this section (for further details see USEPA, 2001a, 2001b).

3.3.1.1 UCM Rounds 1 and 2

The 1987 UCM contaminants include 34 volatile organic compounds (VOCs) (USEPA, 1987; 52 FR 25690). Aldrin and dieldrin, synthetic organic compounds (SOCs), were *not* among these contaminants. The UCM (1987) contaminants were first monitored coincident with the Phase I regulated contaminants, during the 1988-1992 period. This period is often referred to as "Round 1" monitoring. The monitoring data collected by the PWSs were reported to the States (as primacy agents), but there was no protocol in place to report these data to EPA. These data from Round 1 were collected by EPA from many States over time and put into a database called the Unregulated Contaminant Information System, or URCIS.

The 1993 UCM contaminants include 13 SOCs and 1 inorganic contaminant (IOC) (USEPA, 1992; 57 FR 31776). Monitoring for the UCM (1993) contaminants began coincident with the Phase II/V regulated contaminants in 1993 through 1998. This is often referred to as "Round 2" monitoring. The UCM (1987) contaminants were also included in the Round 2 monitoring. As with other monitoring data, PWSs reported these results to the States. EPA, during the past several years, requested that the States submit these historic data to EPA and they are now stored in the SDWIS/FED database.

Monitoring and data collection for aldrin and dieldrin, UCM (1993) contaminants, began in Round 2. Therefore, the following discussion regarding data quality screening, data management, and analytical methods focuses on SDWIS/FED. Discussion of the URCIS

database is included where relevant, but it is worth noting that the various quality screening, data management, and analytical processes were nearly identical for the two databases. For further details on the two monitoring periods as well as the databases, see USEPA (2001a) and USEPA (2001b).

3.3.1.2 Developing a Nationally Representative Perspective

The Round 2 data contain contaminant occurrence data from a total of 35 primacy entities (including 34 States and data for some tribal systems). However, data from some States are incomplete and biased. Furthermore, the national representativeness of the data is problematic because the data were not collected in a systematic or random statistical framework. These State data could be heavily skewed to low-occurrence or high-occurrence settings. Hence, the State data were evaluated based on pollution-potential indicators and the spatial/hydrologic diversity of the nation. This evaluation enabled the construction of a cross-section from the available State data sets that provides a reasonable representation of national occurrence.

A national cross-section comprised of the Round 2 State contaminant occurrence databases was established using the approach developed for the EPA report *A Review of Contaminant Occurrence in Public Water Systems* (USEPA, 1999d). This approach was developed to support occurrence analyses for EPA's Chemical Monitoring Reform (CMR) evaluation, and was supported by peer reviewers and stakeholders. The approach cannot provide a "statistically representative" sample because the original monitoring data were not collected or reported in an appropriate fashion. However, the resultant "national cross-section" of States should provide a clear indication of the central tendency of the national data. The remainder of this section provides a summary description of how the national cross-section from the SDWIS/FED (Round 2) database was developed. The details of the approach are presented in other documents (USEPA, 2001a, 2003b); readers are referred to these for more specific information.

3.3.1.2.1 Cross-Section Development

As a first step in developing the cross-section, the State data contained in the SDWIS/FED database (that contains the Round 2 monitoring results) were evaluated for completeness and quality. Some States reported only detections, or their data had incorrect units. Data sets only including detections are obviously biased, over-representing high-occurrence settings. Other problems included substantially incomplete data sets without all PWSs reporting (USEPA, 2001a; Sections II and III).

The balance of the States remaining after the data quality screening were then examined to establish a national cross-section. This step was based on evaluating the States' pollution potential and geographic coverage in relation to all States. Pollution potential is considered to ensure a selection of States that represent the range of likely contaminant occurrence and a balance with regard to likely high and low occurrence. Geographic consideration is included so that the wide range of climatic and hydrogeologic conditions across the United States are represented, again balancing the varied conditions that affect transport and fate of contaminants, as well as conditions that affect naturally occurring contaminants (USEPA, 2001b; Sections III.A. and III.B.).

The cross-section States were selected to represent a variety of pollution potential conditions. Two primary pollution potential indicators were used. The first factor selected indicates pollution potential from manufacturing/population density and serves as an indicator of the potential for VOC contamination within a State. Agriculture was selected as the second pollution potential indicator because the majority of SOCs of concern are pesticides (USEPA, 2001b; Section III.A.). The 50 individual States were ranked from highest to lowest based on the pollution potential indicator data. For example, the State with the highest ranking for

pollution potential from manufacturing received a ranking of 1 for this factor and the State with the lowest value was ranked as number 50. States were ranked for their agricultural chemical use status in a similar fashion.

The States' pollution potential rankings for each factor were subdivided into four quartiles (from highest to lowest pollution potential). The cross-section States were chosen equally for both pollution potential factors to ensure representation, for example, from: States with high agrochemical pollution potential rankings and high manufacturing pollution potential rankings; States with high agrochemical pollution potential rankings and low manufacturing pollution potential rankings; States with low agrochemical pollution potential rankings and high manufacturing pollution potential rankings; and States with low agrochemical pollution potential rankings and low manufacturing pollution potential rankings (USEPA, 2001b; Section III.B.). In addition, some secondary pollution potential indicators were considered to further ensure that the cross-section States included the spectrum of pollution potential conditions (high to low). At the same time, States within the specific quartiles were considered collectively across all quartiles to attempt to provide a geographic coverage across all regions of the United States.

The data quality screening, pollution potential rankings, and geographic coverage analysis established a national cross-section of 20 Round 2 (SDWIS/FED) States. The cross-section States provide good representation of the nation's varied climatic and hydrogeologic regimes and the breadth of pollution potential for the contaminant groups (Figure 3-1).

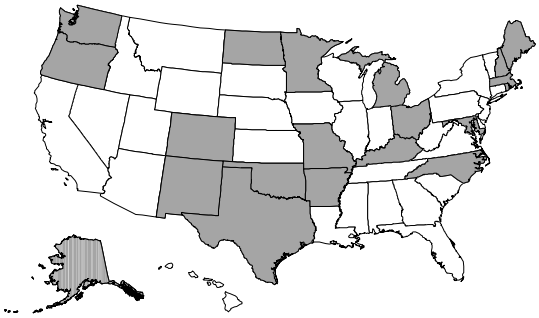
3.3.1.2.2 Cross-Section Evaluation

To evaluate and validate the method for creating the national cross-sections, the method was used to create smaller State subsets from the 24-State, Round 1 (URCIS) cross-section. Again, States were chosen to achieve a balance from the quartiles describing pollution potential, and a balanced geographic distribution, to incrementally build subset cross-sections of various sizes. For example, the Round 1 cross-section was tested with subsets of 4, 8 (the first 4 State subset plus 4 more States), and 13 (8 State subset plus 5) States. Two additional cross-sections were included in the analysis for comparison: a cross-section composed of 16 States with biased data sets eliminated from the 24 State cross-section for data quality reasons, and a cross-section composed of all 40 Round 1 States (USEPA, 2001b; Section III.B.1).

These Round 1 incremental cross-sections were then used to evaluate occurrence for an array of both high and low occurrence contaminants. The comparative results illustrate several points. The results are quite stable and consistent for the 8-, 13- and 24-State cross-sections. They are much less so for the 4-State, 16-State (biased), and 40 State (all Round 1 States) cross-sections. The 4-State cross-section is apparently too small to provide balance both geographically and with pollution potential, a finding that concurs with past work (USEPA, 1999d). The CMR analysis suggested that a minimum of 6-7 States was needed to provide balance both geographically and with pollution potential, and the CMR report used 8 States out of the available data for its nationally representative cross-section (USEPA, 1999d). The 16-State and 40-State cross-sections, both including biased States, provided occurrence results that were unstable and inconsistent for a variety of reasons associated with their data quality problems (USEPA, 2001b; Section III.B.1).

Figure 3-1: Geographic distribution of cross-section States for Round 2 (SDWIS/FED)

Round 2 (SDWIS/FED) Cross Section States	
Alaska	New Hampshire
Arkansas	New Mexico
Colorado	North Carolina
Kentucky	North Dakota
Maine	Ohio
Maryland	Oklahoma
Massachusetts	Oregon
Michigan	Rhode Island
Minnesota	Texas
Missouri	Washington



The 8-, 13-, and 24-State cross-sections provide very comparable results, are consistent, and are usable as national cross-sections to provide estimates of contaminant occurrence. Including greater data from more States improves the national representation and the confidence in the results, as long as the States are balanced related to pollution potential and spatial coverage. The 20-State cross-section provides the best, nationally representative cross-section for the Round 2 data.

3.3.1.3 Data Management and Analysis

The cross-section analyses focused on occurrence at the water system level; i.e., the summary data presented discuss the percentage of public water *systems* with detections, not the percentage of *samples* with detections. By normalizing the analytical data to the system level, skewness inherent in the sample data is avoided. System level analysis was used since a PWS with a known contaminant problem usually has to sample more frequently than a PWS that has never detected the contaminant. Obviously, the results of a simple computation of the percentage of samples with detections (or other statistics) can be skewed by the more frequent sampling results reported by the contaminated site. This level of analysis is conservative. For example, a system need only have a single sample with an analytical result greater than the MRL, i.e., a detection, to be counted as a system with a result "greater than the MRL."

Also, the data used in the analyses were limited to only those data with confirmed water source and sampling type information. Only standard SDWA compliance samples were used; "special" samples, or "investigation" samples (investigating a contaminant problem that would bias results), or samples of unknown type were not used in the analyses. Various quality control and review checks were made of the results, including follow-up questions to the States providing the data. Many of the most intractable data quality problems encountered occurred with older data. These problematic data were, in some cases, simply eliminated from the analysis. For example, when the number of problematic data were insignificant relative to the total number of observations, they were dropped from the analysis (For further details see Cadmus, 2000).

As indicated above, Massachusetts is included in the 20-State, Round 2 national cross-section. Although noteworthy for the presence of SOCs like aldrin and dieldrin, Massachusetts SOC data were problematic. Massachusetts reported Round 2 sample results for SOCs from only 56 PWSs, while reporting VOC results from over 400 different PWSs. Massachusetts SOC

data also contained an atypically high percentage of systems with analytical detections when compared to all other States. Through communications with Massachusetts data management staff it was learned that the State's SOC data were incomplete and that the SDWIS/FED record for Massachusetts SOC data was also incomplete. For instance, the SDWIS/FED Round 2 data for Massachusetts indicate 18% of systems reported detections of aldrin and dieldrin. The States with the next highest detection frequencies reported only 0.2% and 0.4% of systems with detections for aldrin and dieldrin, respectively. In contrast, Massachusetts data characteristics and quantities for IOCs and VOCs were reasonable and comparable with other States' results. Therefore, Massachusetts was included in the group of 20 SDWIS/FED Round 2 cross-section States with usable data for IOCs and VOCs, but its aldrin and dieldrin (SOC) data were omitted from Round 2 cross-section occurrence analyses and summaries presented in this report.

3.3.1.4 Occurrence Analysis

To evaluate national contaminant occurrence, a two-stage analytical approach has been developed. The first stage of analysis provides a straightforward, conservative, non-parametric evaluation of occurrence of the CCL regulatory determination priority contaminants as described above. These Stage 1 descriptive statistics are summarized here. Based in part on the findings of the Stage 1 Analysis, EPA will determine whether more rigorous parametric statistical evaluations, the Stage 2 Analysis, may be warranted to generate national probability estimates of contaminant occurrence and exposure for priority contaminants (for details on this two stage analytical approach see Cadmus, 2000; USEPA, 2002b).

The summary descriptive statistics presented in Table 3-4 for aldrin and in Table 3-5 for dieldrin are a result of the Stage 1 analysis and include data from Round 2 (SDWIS/FED, 1993-1997) cross-section States (excluding Massachusetts). Included are the total number of samples, the percent samples with detections, the 99th percentile concentration of all samples, the 99th percentile concentration of samples with detections, and the median concentration of samples with detections. The percentages of PWSs and population served indicate the proportion of PWSs whose analytical results showed a detection(s) of the contaminant (simple detection, > MRL) at any time during the monitoring period; or a detection(s) greater than half the Health Reference Level (HRL); or a detection(s) greater than the Health Reference Level. The Health Reference Level for aldrin and dieldrin, 0.002 µg/L, is a preliminary estimated health effect level used for this analysis (EPA derived the HRL based on cancer potency selecting the most conservative, and therefore the most protective, value corresponding to the one-in-a-million (1 x 10^{-6}) cancer risk level).

The 99th percentile concentration is used here as a summary statistic to indicate the upper bound of occurrence values because maximum values can be extreme values (outliers) that sometimes result from sampling or reporting error. The 99th percentile concentration is presented for only the samples with detections and for all of the samples because the value for the 99th percentile concentration of all samples is below the Minimum Reporting Level (MRL) (denoted by "<" in Tables 3-4 and 3-5). For the same reason, summary statistics such as the 95th percentile concentration of all samples or the median (or mean) concentration of all samples are omitted because these also are all "<" values. This is the case because only 0.006% (aldrin) and 0.064% (dieldrin) of *all* samples recorded detections in Round 2.

As a simplifying assumption, a value of half the MRL is often used as an estimate of the concentration of a contaminant in samples/systems whose results are less than the MRL. For contaminants with relatively low occurrence such as aldrin and dieldrin in drinking water occurrence databases, the median or mean value of occurrence using this assumption would be half the MRL (0.5 * MRL). However, for these occurrence data this is not straightforward. For Round 2, States have reported a wide range of values for the MRLs. This is in part related to

State data management differences as well as real differences in analytical methods, laboratories, and other factors.

The situation can cause confusion when examining descriptive statistics for occurrence. For example, most Round 2 States reported non-detections simply as zeroes resulting in a modal MRL value of zero. By definition the MRL cannot be zero. This is an artifact of State data management systems. Because a simple meaningful summary statistic is not available to describe the various reported MRLs, and to avoid confusion, MRLs are not reported in the summary tables (Tables 3-4 and 3-5).

In Tables 3-4 and 3-5, national occurrence is estimated by extrapolating the summary statistics for the 20-State cross-section (excluding Massachusetts) to national numbers for systems, and population served by systems, from the *Water Industry Baseline Handbook, Second Edition* (USEPA, 2000d). From the handbook, the total number of community water systems (CWSs) plus non-transient, non-community water systems (NTNCWSs) is 65,030, and the total population served by CWSs plus NTNCWSs is 213,008,182 persons (see Tables 3-4 and 3-5). To generate the estimate of national occurrence based on the cross-section occurrence findings, the national number of PWSs (or population served by PWSs) is simply multiplied by the percentage value for the particular cross section occurrence statistic (e.g. the national estimate for the total number of PWSs with detections (11) is the product of the total national number of PWSs (65,030) and the percentage of PWSs with detections (0.016%)).

Included in Tables 3-4 and 3-5, in addition to the cross-section data results, are results and national extrapolations from all Round 2 reporting States. The data from the biased States are included because of the very low occurrence of aldrin and dieldrin in drinking water samples in all States. For contaminants with very low occurrence, such as aldrin and dieldrin where very few States have detections, any occurrence becomes more important, relatively. For such contaminants, the cross-section process can easily miss a State with occurrence that becomes more important. This is the case with aldrin and dieldrin.

Extrapolating only from the cross-section States, the very low occurrence of aldrin and dieldrin clearly underestimates national occurrence. For example, while data from biased States like Alabama (reporting 100% detections >HRL, >½ HRL, and >MRL for both aldrin and dieldrin) exaggerate occurrence because only systems with detections reported results, their detections are real and need to be accounted for because extrapolations from the cross-section States do not predict enough detections in the biased States. Therefore, results from all reporting Round 2 States, including the biased States, are also used here to extrapolate to a national estimate. Using the biased States' data should provide conservative estimates, likely overestimates, of national occurrence for aldrin and dieldrin.

As exemplified by the cross-section extrapolations for aldrin and dieldrin, national extrapolations of these Stage 1 analytical results can be problematic, especially for contaminants with very low occurrence, because the State data used for the cross-section are not a strict statistical sample. For this reason, the nationally extrapolated estimates of occurrence based on Stage 1 results are not presented in the *Federal Register* Notice. The presentation in the *Federal Register* Notice of only the actual results of the cross-section analysis maintains a straight-forward presentation, and the integrity of the data, for stakeholder review. The nationally extrapolated Stage 1 occurrence values are presented here, however, to provide additional perspective. A more rigorous statistical modeling effort, the Stage 2 analysis, could be conducted on the cross-section data (USEPA, 2002b). The Stage 2 results would be more statistically robust and more suitable to national extrapolation. This approach would provide a probability estimate and would also allow for better quantification of estimation error.

3.3.1.5 Additional Drinking Water Data from the Corn Belt

To augment the SDWA drinking water data analysis described above, and to provide additional coverage of the corn belt States where the use of aldrin and dieldrin as agricultural insecticides was historically high, independent analyses of SDWA drinking water data from the States of Iowa, Illinois, and Indiana are reviewed below. Raw water aldrin and dieldrin monitoring data are also included from rural, private water supply wells in Illinois.

The Iowa analysis examined SDWA compliance monitoring data from surface and ground water PWSs for the years 1988-1995 (Hallberg et al., 1996). Illinois and Indiana compliance monitoring data for surface and ground water PWSs were evaluated mostly for the years after 1993, though some earlier data were also included (USEPA, 1999d). The raw water data from Illinois were collected from rural, private supply wells (Goetsch et al., 1992). Data sources, data quality, and analytical methods for these analyses are described in the respective reports; they were all treated similarly to the data quality reviews for this analysis.

3.3.2 Results

3.3.2.1 Aldrin

3.3.2.1.1 Occurrence Estimates

The percentages of PWSs with detections of aldrin are very low (Table 3-4). The cross-section shows only approximately 0.02% of PWSs (approximately 11 PWSs nationally) experienced detections at any concentration level (>MRL, >½ HRL, and >HRL), affecting about 0.02% of the population served (approximately 40,000-50,000 people nationally) (see also Figure 3-3). All of the detections were in systems using ground water. The percentage of PWSs (or population served) in a given source category (i.e., ground water) with detections >MRL, >½ HRL, or >HRL is the same because the estimated HRL is lower than the MRL. Hence, any detection reported is also greater than the HRL. While concentrations are low — for the detections the median concentration is 0.58 µg/L and the 99th percentile concentration is 0.69 µg/L — these values are greater than the HRL.

As noted above, because of the very low occurrence, the cross-section States yield an underestimate. Hence, all data are used, even the biased data, to present a conservative upper bound estimate. Conservative estimates of aldrin occurrence using all of the Round 2 reporting States still show relatively low detection frequencies (Table 3-4). Approximately 0.2% of PWSs (estimated at 138 PWSs nationally) experienced detections at any concentration level (>MRL, >½ HRL, and >HRL), affecting about 0.5% of the population served (approximately 1,052,000 people nationally). The proportion of surface water PWSs with detections was greater than ground water systems. Again the percentages of PWSs (or populations served) with detections >MRL, >½ HRL, or >HRL are the same because of the low HRL. The median concentration of detections is 0.18 µg/L and the 99th percentile concentration is 4.4 µg/L.

The Round 2 reporting States and the Round 2 national cross-section show a proportionate balance in PWS source waters compared to the national inventory. Nationally, 91% of PWSs use ground water (and 9% surface waters): Round 2 reporting States and the Round 2 national cross-section show that 87% use ground water (and 13% surface waters). The relative populations served are not as comparable. Nationally, about 40% of the population is served by PWSs using ground water (and 60% by surface water). For the Round 2 cross-section, 29% of the cross-section population is served by ground water PWSs (and 71% by surface water). For all Round 2 reporting States, 31% of the population is served by ground water PWSs (and 69% by surface water). As a consequence of these disproportions, the resultant national extrapolations are not additive.

Drinking water data from the corn belt States of Iowa, Indiana, and Illinois also show very low occurrence of aldrin. There were no detections of the pesticide in the Iowa or Indiana SDWA compliance monitoring data for surface or ground water PWSs (Hallberg et al., 1996; USEPA, 1999d). While Illinois also had no detections of the compound in ground water PWSs, it was detected in surface water PWSs in that State. Occurrence was low with 1.8% of surface water systems, and 0.10% of samples, showing detections. The 99th percentile concentration of all samples was below the reporting level and the maximum concentration was 2.4 µg/L (USEPA, 1999d). A survey of Illinois rural, private water supply wells showed very low occurrence of aldrin as well. Only 0.3% of all sampled wells had detections at a reporting limit of 0.004 µg/L (Goetsch et al., 1992).

Table 3-4: Summary occurrence statistics for aldrin

Frequency Factors	20 State Cross-Section[1] (Round 2)	All Reporting States[2] (Round 2)	National System & Population Numbers[3]	
Total Number of Samples	31,083	41,565	--	
Percent of Samples with Detections	0.006%	0.132%	--	
99th Percentile Concentration (all samples)	< (Non-detect)	< (Non-detect)	--	
Health Reference Level	0.002 µg/L	0.002 µg/L	--	
Minimum Reporting Level (MRL)	Variable[4]	Variable[4]	--	
99th Percentile Concentration of Detections	0.69 µg/L	4.40 µg/L	--	
Median Concentration of Detections	0.58 µg/L	0.18 µg/L	--	
Total Number of PWSs	12,165	15,123	65,030	
Number of GW PWSs	10,540	13,195	59,440	
Number of SW PWSs	1,625	1,928	5,590	
Total Population	47,708,156	58,979,361	213,008,182	
Population of GW PWSs	14,043,051	18,279,343	85,681,696	
Population of SW PWSs	33,665,105	40,700,018	127,326,486	
Occurrence by System			**National Extrapolation[5]**	
% PWSs with detections (> MRL)	0.016%	0.212%	11	138
Range of Cross-Section States	0 - 0.23%	0 - 100 %	N/A	N/A
GW PWSs with detections	0.019%	0.167%	11	99
SW PWSs with detections	0.000%	0.519%	0	29
% PWSs > 1/2 Health Reference Level (HRL)	0.016%	0.212%	11	138
Range of Cross-Section States	0 - 0.23%	0 - 100 %	N/A	N/A
GW PWSs > 1/2 Health Reference Level	0.019%	0.167%	11	99
SW PWSs > 1/2 Health Reference Level	0.000%	0.519%	0	29
% PWSs > Health Reference Level	0.016%	0.212%	11	138
Range of Cross-Section States	0 - 0.23%	0 - 100 %	N/A	N/A
GW PWSs > Health Reference Level	0.019%	0.167%	11	99
SW PWSs > Health Reference Level	0.000%	0.519%	0	29
Occurrence by Population Served				
% PWS Population Served with detections	0.018%	0.494%	39,000	1,052,000
Range of Cross-Section States	0 - 0.35%	0 - 100 %	N/A	N/A
GW PWS Population with detections	0.062%	0.414%	53,000	355,000
SW PWS Population with detections	0.000%	0.530%	0	674,000
% PWS Population Served > 1/2 Health Reference Level	0.018%	0.494%	39,000	1,052,000
Range of Cross-Section States	0 - 0.35%	0 - 100 %	N/A	N/A
GW PWS Population > 1/2 Health Reference Level	0.062%	0.414%	53,000	355,000
SW PWS Population > 1/2 Health Reference Level	0.000%	0.530%	0	674,000
% PWS Population Served > Health Reference Level	0.018%	0.494%	39,000	1,052,000
Range of Cross-Section States	0 - 0.35%	0 - 100 %	N/A	N/A
GW PWS Population > Health Reference Level	0.062%	0.414%	53,000	355,000
SW PWS Population > Health Reference Level	0.000%	0.530%	0	674,000

1. Summary Results based on data from 20-State Cross-Section (minus Massachusetts), from SDWIS/FED, UCM (1993) Round 2.
2. Summary Results based on data from all reporting States from SDWIS/FED, UCM (1993) Round 2.
3. Total PWS and population numbers are from EPA March 2000 Water Industry Baseline Handbook.
4. See Section 3.3.1.4 for discussion.
5. National extrapolations are from the 20-State cross-section data (left) and all Round 2 States reporting data (right) using the Baseline Handbook system and population numbers.
- PWS = Public Water Systems; GW = Ground Water; SW = Surface Water; MRL = Minimum Reporting Level (for laboratory analyses);
HRL = Health Reference Level, an estimated health effect level used for preliminary assessment for this review; N/A = Not Applicable."
- The Health Reference Level (HRL) used for aldrin is 0.002 µg/L. This is a draft value for working review only.
- Total Number of Samples = the total number of analytical records for aldrin.
- 99th Percentile Concentration = the concentration value of the 99th percentile of either all analytical results or just the detections (in µg/L)
- Median Concentration of Detections = the median analytical value of all the detections (analytical results greater than the MRL) (in µg/L).
- Total Number of PWSs = the total number of public water systems with records for aldrin
- Total Population Served = the total population served by public water systems with records for aldrin
- % PWS with detections, % PWS > ½ Health Reference Level, % PWS > Health Reference Level = percent of the total number of public water systems with at least one analytical result that exceeded the MRL, ½ Health Reference Level, Health Reference Level, respectively
- % PWS Population Served with detections, % PWS Population Served >½ Health Reference Level, % PWS Population Served > Health Reference Level = percent of the total population served by PWSs with at least one analytical result exceeding the MRL, ½ Health Reference Level, or the Health Reference Level, respectively

Figure 3-2: States with PWSs with detections of aldrin for all States with data in SDWIS/FED (Round 2)

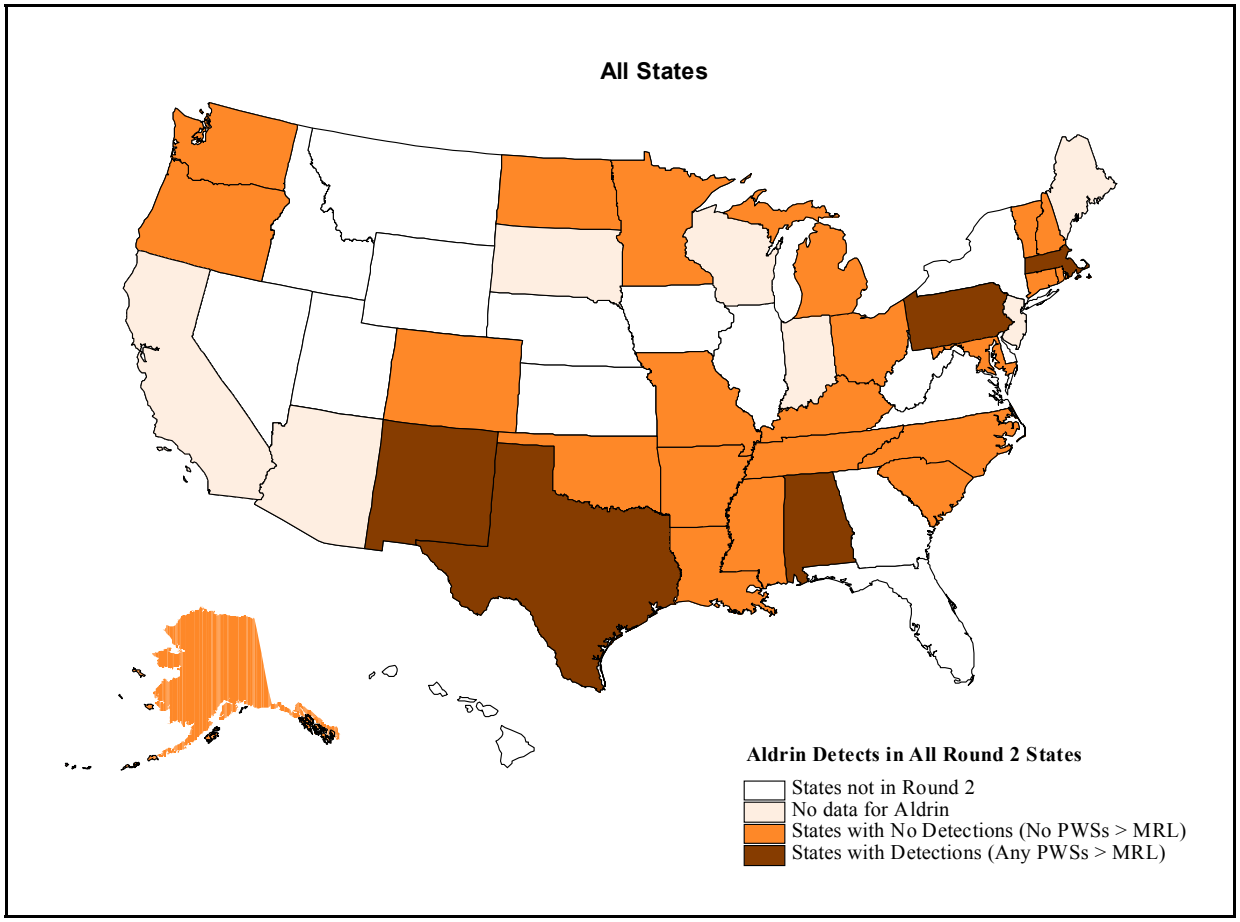
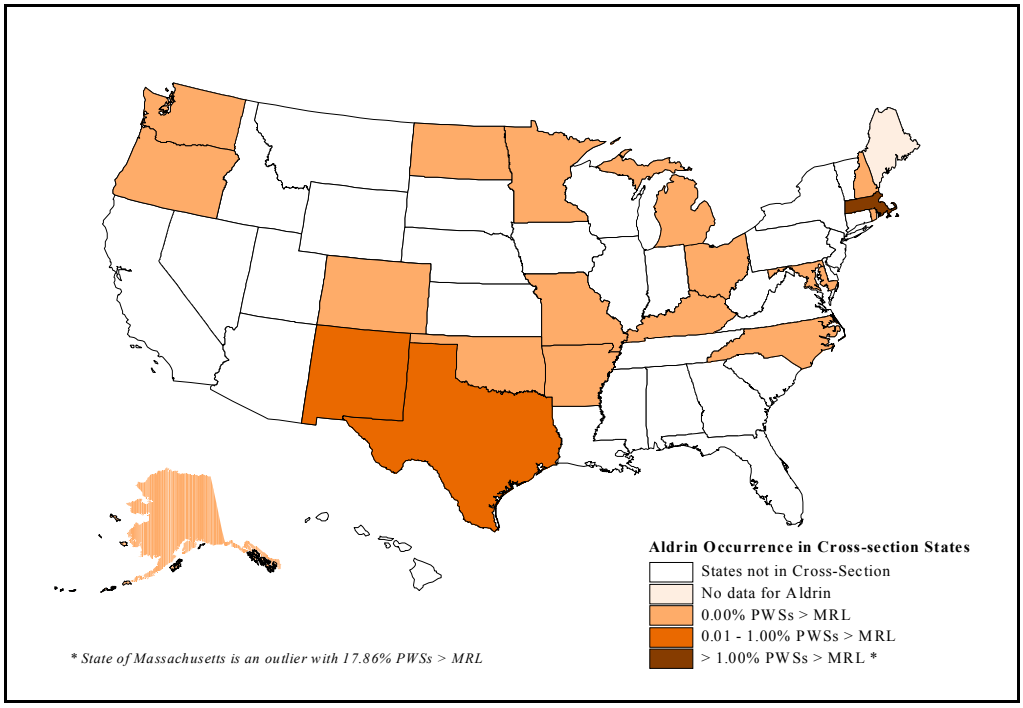
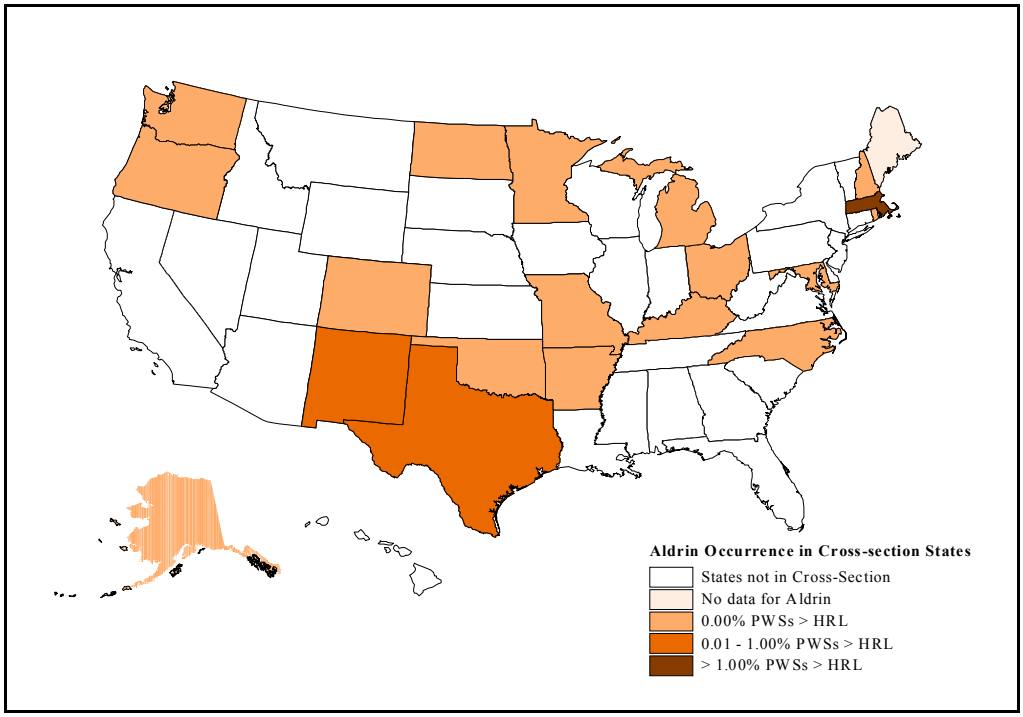


Figure 3-3: Round 2 cross-section States with PWSs with detections of aldrin (any PWSs with results greater than the Minimum Reporting Level [MRL]; above) and concentrations greater than the Health Reference Level (HRL; below)





3.3.2.1.2 Regional Patterns

Occurrence results are displayed graphically by State in Figures 3-2 and 3-3 to assess whether any distinct regional patterns of occurrence are present. Thirty-four States reported Round 2 data but seven of those States have no data for aldrin (Figure 3-2). Another 22 States did not detect aldrin. The remaining 5 States have detected aldrin in drinking water and are generally located either in the southern United States or the Northeast (Figure 3-2). In contrast to the summary statistical data presented in the previous section, this simple spatial analysis includes the biased Massachusetts data.

The simple spatial analysis presented in Figures 3-2 and 3-3 suggests that special regional analyses are not warranted. The State of Alabama does, however, stand out as having relatively high occurrence for reasons that are unclear. While there is a weak geographic clustering of drinking water detections in a few southern and northeastern States (including the State of Massachusetts' biased data), this is partly the result of so few States with any detections. Further, use and environmental release information described in section 3.1 of this report indicates that aldrin detections are more widespread than the drinking water data suggest. Two out of the three TRI States (Arkansas and Michigan) that reported releases of aldrin into the environment did not report detections of the chemical in PWS sampling. Furthermore, aldrin's widespread presence in the environment is evidenced by detections of the compound in hazardous waste sites in at least 31 States (at NPL sites), as well as detections in site samples in at least 40 States (listed in ATSDR's HazDat).

3.3.2.2 Dieldrin

3.3.2.2.1 Occurrence Estimates

The percentages of PWSs with detections are very low (Table 3-5). The cross-section shows approximately 0.1% of PWSs (about 61 PWSs nationally) experienced detections at any concentration level (>MRL, >½ HRL, and >HRL), affecting less than 0.1% of the population served (approximately 150,000 people nationally) (see also Figure 3-5). The percentages of PWSs (or population served) in a given source category (i.e., ground water) with detections greater than the MRL, ½ HRL, or HRL are the same because the estimated HRL is less than the MRL. Hence, any detection reported is also greater than the HRL. Detection frequencies are marginally higher for surface water systems when compared to ground water systems. While concentrations are also low — for samples with detections the median concentration is 0.16 µg/L and the 99th percentile concentration is 1.36 µg/L — these values are greater than the HRL.

As noted above, because of the very low occurrence, the cross-section States yield an underestimate. Hence, all data are used, even the biased data, to present a conservative upper bound estimate. Conservative estimates of dieldrin occurrence using all of the Round 2 reporting States still show relatively low detection frequencies (Table 3-5). Approximately 0.2% of PWSs (estimated at 137 PWSs nationally) experienced detections at any concentration level (>MRL, >½ HRL, and >HRL), affecting about 0.4% of the population served (approximately 793,000 people nationally). The proportion of surface water PWSs with detections was greater than ground water systems. Again the percentages of PWSs (or populations served) with detections greater than the MRL, ½ HRL, or HRL are the same because of the low HRL. The median concentration of detections is 0.42 µg/L and the 99th percentile concentration is 4.4 µg/L.

The Round 2 reporting States and the Round 2 national cross-section show a proportionate balance in PWS source waters compared to the national inventory. Nationally, 91% of PWSs use
ground water (and 9% surface waters): Round 2 reporting States and the Round 2 national cross-section show that 88% use ground water (and 12% surface waters). The relative populations

served are not similarly comparable. Nationally, about 40% of the population is served by PWSs using ground water (and 60% by surface water). For the Round 2 cross-section, 30% of the cross-section population is served by ground water PWSs (and 70% by surface water). For all Round 2 reporting States, 32% of the population is served by ground water PWSs (and 68% by surface water). As a consequence of these disproportions, the resultant national extrapolations are not additive.

Drinking water data from the corn belt States of Iowa, Indiana, and Illinois also show very low occurrence of dieldrin. There were no detections of the pesticide in the Iowa SDWA compliance monitoring data for surface or ground water PWSs (Hallberg et al, 1996). While Illinois and Indiana also had no detections of the compound in ground water PWSs, it was detected in surface water PWSs in those States (USEPA, 1999d). Occurrence was low in both States: 1.8% of surface water systems (0.1% of samples) showed detections in Illinois and 2.1% of surface water systems (0.3% of samples) showed detections in Indiana. For Illinois and Indiana surface water PWSs, the 99th percentile concentrations of all samples were below the reporting level and the maximum concentrations were 0.1 µg/L and 0.04 µg/L, respectively (USEPA, 1999d). Furthermore, in a survey of rural, private water supply wells in Illinois, only 1.6% of all sampled wells had detections of dieldrin (Goetsch et al., 1992).

3.3.2.2.2 Regional Patterns

Occurrence results are displayed graphically by State in Figures 3-4 and 3-5 to assess whether any distinct regional patterns of occurrence are present. Thirty-four States reported Round 2 data but 7 of those States have no data for dieldrin (Figure 3-4). Another 19 States did not detect dieldrin. The remaining 8 States detected dieldrin in drinking water, and are generally located either in the southern United States or in the Northeast (Figure 3-4). In contrast to the summary statistical data presented in the previous section, this simple spatial analysis includes the biased Massachusetts data.

The simple spatial analysis presented in Figures 3-4 and 3-5 suggests that special regional analyses are not warranted. The State of Alabama does, however, stand out as having relatively high occurrence for reasons that are unclear. While there is a weak geographic clustering of drinking water detections in a few southern and northeastern States (including the State of Massachusetts' biased data), this is partly the result of so few States with any detections. Further, use and environmental release information (section 3.1) and ambient water quality data (section 3.2) indicate that dieldrin detections are more widespread than the drinking water data suggest. Detections of the compound in hazardous waste sites in at least 38 States (at NPL sites), site samples in at least 40 States (listed in ATSDR's HazDat), and water, sediment, and biotic tissue quality data from the NAWQA program provide evidence for nationwide occurrence.

3.4 Conclusion

Aldrin has been detected at very low frequencies and concentrations in bed sediments sampled during the first round of the USGS NAWQA studies. Aldrin releases have been reported through TRI. Dieldrin has been detected at low frequencies and concentrations in ground and surface water sampled during the first round of the USGS NAWQA studies, and at similar frequencies and concentrations in surface waters of the Mississippi River and major tributaries. Dieldrin occurrence is greater in stream bed sediments and biotic tissue, however. Both aldrin and dieldrin have been found at ATSDR HazDat and CERCLA NPL sites across the country.

Aldrin and dieldrin have been detected in PWS samples collected under SDWA. Occurrence estimates are very low with only 0.006% and 0.06% of all cross-section samples showing

detections for aldrin and dieldrin, respectively. Significantly, the values for the 99th percentile and median concentrations of all cross-section samples are less than the MRL for both contaminants. For Round 2 cross-section samples with detections, the median concentration for aldrin is 0.58 µg/L and the 99th percentile concentration is 0.69 µg/L. Dieldrin Round 2 cross-section samples with detections show median and 99th percentile concentrations of 0.16 µg/L and 1.36 µg/L, respectively. Systems with detections constitute only 0.02% of Round 2 cross-section systems (an estimate of 11 systems, nationally) for aldrin, and 0.1% of Round 2 cross-section systems (about 61 systems, nationally) for dieldrin.

National estimates for the population served by PWSs with detections are also very low (40,000-50,000 for aldrin and 150,000 for dieldrin), and are the same for all categories (>MRL, >½ HRL, >HRL). These estimates constitute less than 0.02% of the national population for aldrin and less than 0.1% of the national population for dieldrin. Using more conservative estimates of occurrence from all States reporting SDWA Round 2 monitoring data, including States with biased data, 0.2% of the nation's PWSs (approximately 138 systems) and 0.5% of the PWS population served (approximately 1,052,000 people) are estimated to have aldrin detections greater than the MRL, ½ HRL, and HRL. More conservative estimates for dieldrin show that 0.2% of the nation's PWSs (approximately 137 systems) and 0.4% of the PWS population served (approximately 793,000 people) are estimated to have detections of dieldrin greater than the MRL, half the HRL, and the HRL.

Additional SDWA compliance data from the corn belt States of Iowa, Indiana, and Illinois examined through independent analyses support the drinking water data analyzed in this report. There were no detections of aldrin or dieldrin in either surface or ground water PWSs in Iowa. Illinois reported aldrin and dieldrin detections from surface water PWSs only, with 1.8% of surface water systems (0.1% of samples) showing aldrin occurrence and 1.8% of surface water systems (0.1% of samples) showing dieldrin occurrence. Indiana reported no detections of aldrin in either surface or ground water, though 2.1% of surface water systems (0.3% of samples) reported dieldrin detections. The 99th percentile concentration of all aldrin samples was below the reporting level and the maximum concentration was 2.4 µg/L. The 99th percentile concentrations of all dieldrin samples from Illinois and Indiana were also below the reporting level while the maximum concentrations were 0.1 µg/L and 0.04 µg/L, respectively (USEPA, 1999d). Further, in a survey of rural, private water supply wells in Illinois, aldrin and dieldrin were detected in only 0.3% and 1.6% of all sampled wells, respectively.

4.0 HEALTH EFFECTS

A full description of the health effects and the dose-response information for threshold and non-threshold effects associated with exposures to aldrin or dieldrin are presented in Chapters 7 and 8 of the Drinking Water Support Document for Aldrin and Dieldrin (USEPA, 2003b). A summary of the pertinent findings are presented below.

4.1 Hazard Characterization and Mode of Action Implications

Following acute exposure to high doses, the primary adverse health effects of aldrin and dieldrin in humans are those resulting from neurotoxicity to the central nervous system, including hyperirritability, convulsions and coma (Jager, 1970; Spiotta, 1951; ACGIH, 1984). In some cases, these may be followed by cardiovascular effects such as unusually rapid beating of the heart and elevated blood

Table 3-5: Summary occurrence statistics for dieldrin

Frequency Factors	20 State Cross-Section[1] (Round 2)	All Reporting States[2] (Round 2)	National System & Population Numbers[3]	
Total Number of Samples	29,603	40,055	--	
Percent of Samples with Detections	0.064%	0.135%	--	
99th Percentile Concentration (all samples)	< (Non-detect)	< (Non-detect)	--	
Health Reference Level	0.002 µg/L	0.002 µg/L	--	
Minimum Reporting Level (MRL)	Variable[4]	Variable[4]	--	
99th Percentile Concentration of Detections	1.36 µg/L	4.40 µg/L	--	
Median Concentration of Detections	0.16 µg/L	0.42 µg/L	--	
Total Number of PWSs	11,788	14,725	65,030	
Number of GW PWSs	10,329	12,968	59,440	
Number of SW PWSs	1,459	1,757	5,590	
Total Population	45,784,187	56,909,027	213,008,182	
Population of GW PWSs	13,831,864	18,044,000	85,681,696	
Population of SW PWSs	31,952,323	38,865,027	127,326,486	
Occurrence by System			**National Extrapolation[5]**	
% PWSs with detections (> MRL)	0.093%	0.211%	61	137
Range of Cross-Section States	0 - 0.97%	0 - 100%	N/A	N/A
GW PWSs with detections	0.087%	0.177%	52	105
SW PWSs with detections	0.137%	0.455%	8	25
% PWSs > 1/2 Health Reference Level (HRL)	0.093%	0.211%	61	137
Range of Cross-Section States	0 - 0.97%	0 - 100%	N/A	N/A
GW PWSs > 1/2 Health Reference Level	0.087%	0.177%	52	105
SW PWSs > 1/2 Health Reference Level	0.137%	0.455%	8	25
% PWSs > Health Reference Level	0.093%	0.211%	61	137
Range of Cross-Section States	0 - 0.97%	0 - 100%	N/A	N/A
GW PWSs > Health Reference Level	0.087%	0.177%	52	105
SW PWSs > Health Reference Level	0.137%	0.455%	8	25
Occurrence by Population Served				
% PWS Population Served with detections	0.070%	0.372%	150,000	793,000
Range of Cross-Section States	0 - 2.00%	0 - 100%	N/A	N/A
GW PWS Population with detections	0.146%	0.371%	125,000	318,000
SW PWS Population with detections	0.038%	0.372%	48,000	474,000
% PWS Population Served > 1/2 Health Reference Level	0.070%	0.372%	150,000	793,000
Range of Cross-Section States	0 - 2.00%	0 - 100%	N/A	N/A
GW PWS Population > 1/2 Health Reference Level	0.146%	0.371%	125,000	318,000
SW PWS Population > 1/2 Health Reference Level	0.038%	0.372%	48,000	474,000
% PWS Population Served > Health Reference Level	0.070%	0.372%	150,000	793,000
Range of Cross-Section States	0 - 2.00%	0 - 100%	N/A	N/A
GW PWS Population > Health Reference Level	0.146%	0.371%	125,000	318,000
SW PWS Population > Health Reference Level	0.038%	0.372%	48,000	474,000

1. Summary Results based on data from 20-State Cross-Section (minus Massachusetts), from SDWIS/FED, UCM (1993) Round 2.
2. Summary Results based on data from all reporting States from SDWIS/FED, UCM (1993) Round 2; see text for further discussion.
3. Total PWS and population numbers are from EPA March 2000 Water Industry Baseline Handbook.
4. See Section 3.3.1.4 for discussion
5. National extrapolations are from the 20-State cross-section data (left) and all Round 2 States reporting data (right) using the Baseline Handbook system and population numbers.
- "PWS = Public Water Systems; GW = Ground Water; SW = Surface Water; MRL = Minimum Reporting Level (for laboratory analyses); HRL= Health Reference Level, an estimated health effect level used for preliminary assessment for this review; N/A = Not Applicable"
- The Health Reference Level (HRL) used for dieldrin is 0.002 µg/L. This is a draft value for working review only.
- Total Number of Samples = the total number of analytical records for dieldrin
- 99th Percentile Concentration = the concentration value of the 99th percentile of either all analytical results or just the detections (in µg/L)
- Median Concentration of Detections = the median analytical value of all the detections (analytical results greater than the MRL) (in µg/L)
- Total Number of PWSs = the total number of public water systems with records for dieldrin
- Total Population Served = the total population served by public water systems with records for dieldrin
- % PWS with detections, % PWS > ½ Health Reference Level, % PWS > Health Reference Level = percent of the total number of public water systems with at least one analytical result that exceeded the MRL, ½ Health Reference Level, Health Reference Level, respectively
- % PWS Population Served with detections, % PWS Population Served >½ Health Reference Level, % PWS Population Served > Health Reference Level = percent of the total population served by PWSs with at least one analytical result exceeding the MRL, ½ Health Reference Level, or the Health Reference Level, respectively

Figure 3-4: States with PWSs with detections of dieldrin for all States with data in SDWIS/FED (Round 2)

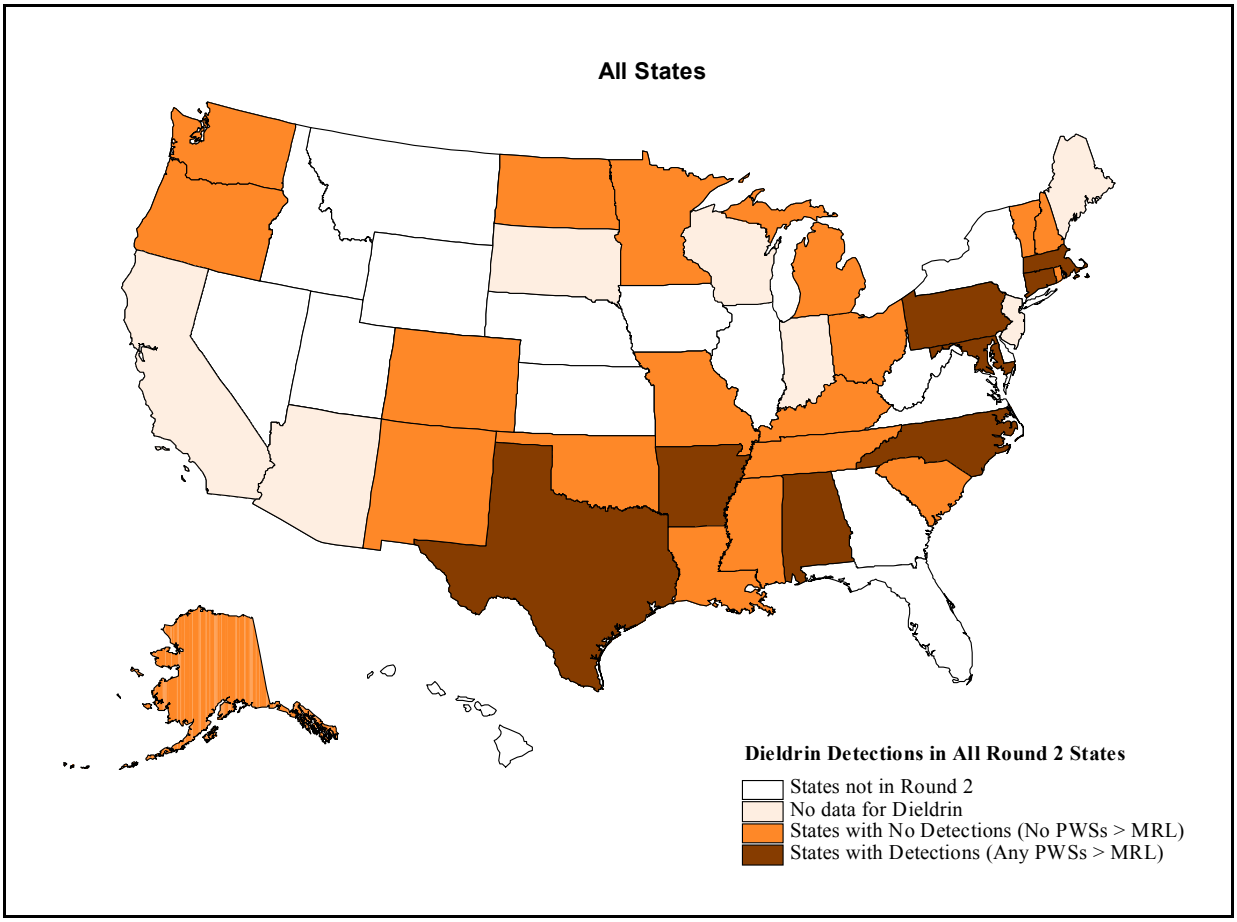
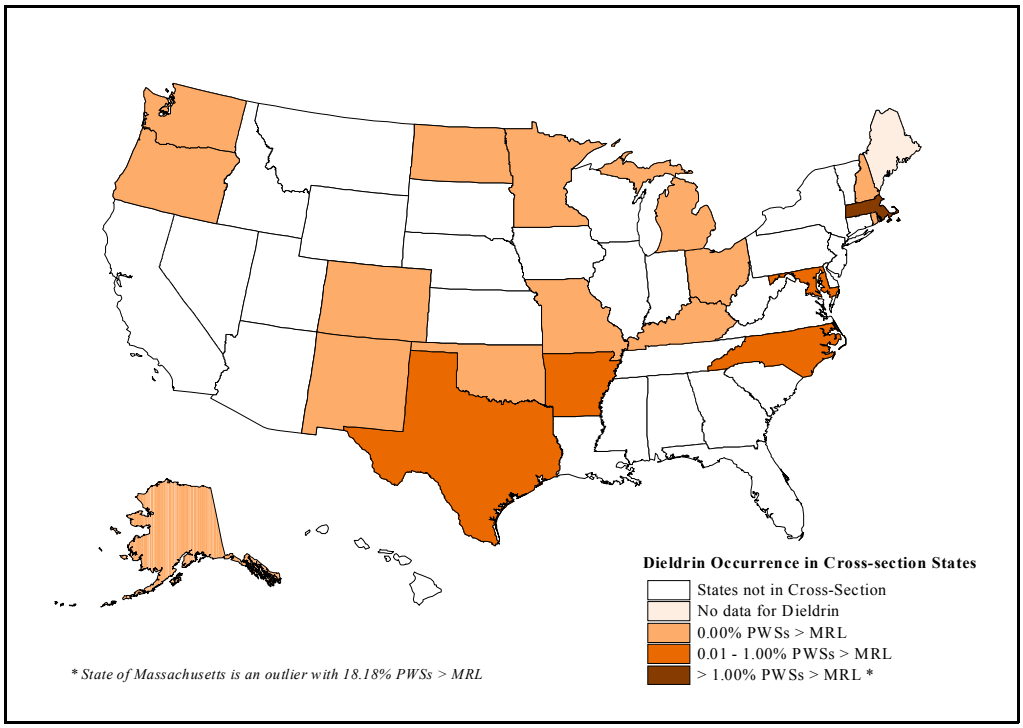
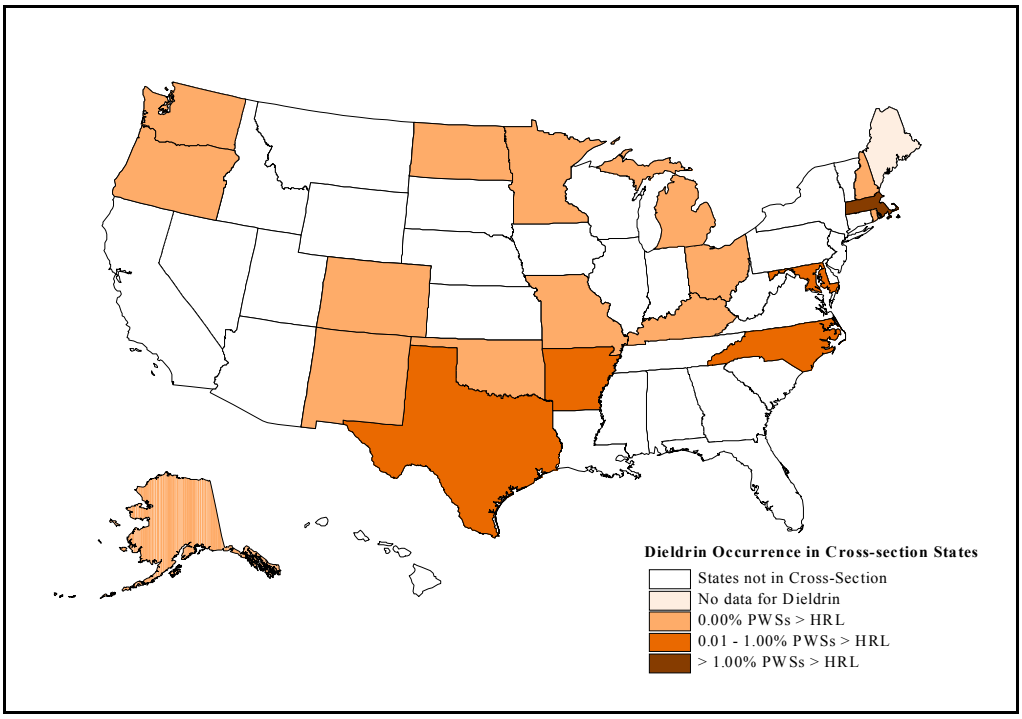


Figure 3-5: Round 2 cross-section States with PWSs with detections of dieldrin (any PWSs with results greater than the Minimum Reporting Level [MRL]; above) and concentrations greater than the Health Reference Level (HRL; below)





pressure (Black, 1974). Under conditions of longer-term exposure to lower doses of these compounds, neurotoxic symptoms may also include headache, dizziness, general malaise, nausea, vomiting, and muscle twitching or muscle spasms (Jager, 1970; ATSDR, 2000b). Dieldrin exposure has been linked to two cases of immunohemolytic anemia, where an immune system response destroys red blood cells (Hamilton et al., 1978; Muirhead et al., 1959), and aldrin/dieldrin exposure has been linked with several instances of aplastic anemia, where bone marrow cannot adequately regenerate red blood cells (de Jong, 1991; Pick et al., 1965; ATSDR, 2000b). However, at least some of these studies are problematic. In any case, hematological or immunological (e.g., dermal sensitization) effects have not generally been found in humans following exposure to either compound.

Common acute or subchronic neurotoxic effects observed in animals are characterized by increased irritability, salivation, hyperexcitability, tremors followed by convulsions, loss of body weight, depression, prostration, and death (Borgmann et al., 1952; Walker et al., 1969; Wagner and Greene, 1978; Woolley et al., 1985; NCI, 1978; Casteel, 1993). Manifestations of hepatotoxicity (e.g., elevated serum enzyme levels, unregulated liver cell regenerations, rapid turn-over in bile duct cells, localized cell degeneration and death, etc.) have been observed in animals following subchronic-to-chronic exposure to moderate-to-high concentrations of aldrin/dieldrin (summarized in ATSDR, 2000b). Relatively low-dose, chronic exposures to either compound have been associated with changes in the tissue structure of the liver in rat studies (e.g., Fitzhugh et al., 1964; Walker et al., 1969). There is some evidence from animals that aldrin/dieldrin exposure may either induce renal lesions or exacerbate pre-existing kidney illness (ATSDR, 2000b; Fitzhugh et al., 1964; Harr et al., 1970).

Various *in vivo* and *in vitro* studies have provided evidence that aldrin and dieldrin may act as weak endocrine disruptors. Changes in male and female hormone levels and/or receptor binding, and degeneration of male germ cells and seminiferous tubules in the testes (the Leydig cell ultrastructure) have been observed. In females, effects on the estrus cycle and rapid turnover of breast cells and of the cells lining the uterus have been reported (ATSDR, 2000b). Oral administration of aldrin/dieldrin to maternal or paternal animals has produced somewhat equivocal evidence of decreased fertility (Dean et al., 1975; Epstein et al., 1972; Good and Ware, 1969; Harr et al., 1970; Virgo and Bellward, 1975), and injection of aldrin in the abdominal cavity has produced various adverse effects on the male reproductive system (ATSDR, 2000b).

Immunosuppression by dieldrin has been reported in a number of mouse studies (Krzystyniak et al., 1985; Loose, 1982; Loose et al., 1981). A number of long-term bioassay studies have provided evidence that aldrin and dieldrin also cause liver cancer in mice (Davis and Fitzhugh, 1962; Davis, 1965; Song and Harville, 1964; NCI, 1978; MacDonald et al., 1972; Walker et al., 1972; Thorpe and Walker, 1973; Meierhenry et al., 1983). In one mouse study, dieldrin was found to have induced lung, lymphoid and "other" tumors (Walker et al., 1972). In contrast, neither compound has been found to induce liver tumors in rats (Treon and Cleveland, 1955; Song and Harville, 1964; Deichmann et al., 1967, 1970; Deichmann, 1974; NCI, 1978; Fitzhugh et al., 1964; Walker et al., 1969), although a number of these studies suffered from one or more serious deficiencies.

Despite some sporadic, statistically significant increases in rectal or liver/biliary cancer, occupational and epidemiological studies have failed to provide any convincing evidence for the carcinogenicity of either aldrin or dieldrin in humans (Van Raalte, 1977; Versteeg and Jager, 1973; de Jong, 1991; de Jong et al., 1997; Ditraglia et al., 1981; Brown, 1992; Amaoteng-Adjepong et al., 1995). In fact, the ratio of deaths in the exposed vs. general populations for both specific causes and all causes of death have generally been less than 1.0.

Not a great deal is known about the modes of action that may underlie the various toxic effects produced by exposure to aldrin or dieldrin. The hyperexcitability associated with these

compounds' neurotoxicity has generally been thought to arise from increased and unregulated nerve impulses throughout the central nervous system, but whether this results from facilitated neurotransmitter release at the nerve terminals, or from reducing the activity of inhibitory neurotransmitters within the central nervous system, has been unclear (ATSDR, 2000b). Mehrota et al. (1988, 1989) have proposed that dieldrin may act by inhibiting a calcium-dependent brain enzyme (ATPase), resulting in higher intracellular calcium levels that would promote neurotransmitter release. More recent work provides significant evidence that aldrin and dieldrin's principal mode of neurotoxic action likely involves their role as antagonists for the membrane receptor for the inhibitory neurotransmitter, gamma aminobutyric acid (GABA), blocking the influx of chloride ion through the GABA-controlled channel (Klaassen, 1996; Nagata and Narahashi, 1994, 1995; Nagata et al., 1994; Brannen et al., 1998; Johns et al., 1998; Liu et al., 1997, 1998). Additionally, at least one *in vitro* study using fetal rat brain cells suggests that dieldrin may have an even greater functional effect on the nerve cells that use dopamine as a neurotransmitter (Sanchez-Ramos et al., 1998).

Genotoxicity, defined as direct action of the chemical on the genetic material (i.e., DNA), is not expected to have a predominant role in the mode of action for these compounds' carcinogenicity. The capacity of aldrin and dieldrin to inhibit various forms of *in vitro* intercellular communication in both human and animal cells may be significant in triggering tumor production due to cell proliferation, unregulated cell growth and premature cellular death (Kurata et al., 1982; Wade et al., 1986; Zhong-Xiang et al., 1986; Mikalsen and Sanner, 1993). However, these growth effects are epigenetic: they are unrelated to any change in DNA. Generally speaking, hepatocarcinogenic effects in mice may potentially be associated with epigenetic modes of action, but not in rats.

4.2 Dose-Response Characterization and Implications in Risk Assessment

In adult humans, the acute oral lethal dose for aldrin/dieldrin has been estimated at approximately 70 mg/kg bw (Jager, 1970; ATSDR, 2000b), which is about 3 times the dose reported to have induced convulsions within 20 minutes of ingestion (Spiotta, 1951). The oral dose that is sufficient to kill 50 percent of the treated animals (Oral LD_{50}) was determined in various animal species for the two compounds; these values have been reported to range from 33-95 mg/kg bw, and appear to be affected by age at the time of exposure. In rats, LD_{50} values were reported as 37 mg/kg bw for young adults, 25 mg/kg bw for two week-old pups, and a high 168 mg/kg bw for newborns (Lu et al., 1965).

Meaningful dose-response relationships have not been adequately characterized in humans for any of the toxic effects of aldrin or dieldrin. In animals, oral exposure to aldrin/dieldrin has produced a variety of dose-dependent effects over a collective dose range of at least three orders of magnitude (< 0.05-50 mg/kg bw), depending on the specific endpoint and the duration of exposure (ATSDR, 2000b). Dose-response information for some key chronic and cancer bioassay studies are summarized in Table 9-1 of the Drinking Water Support Document for Aldrin and Dieldrin (USEPA, 2001). For noncancer effects, the USEPA has determined oral Reference Doses (RfDs) for both aldrin and dieldrin based upon the most sensitive relevant toxic effects (critical effects) that have been reported. The RfD is an estimate, with an uncertainty spanning perhaps an order of magnitude, of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. For aldrin, the critical effect was liver toxicity observed in rats after chronic exposure to approximately 0.025 mg/kg bw-day, the Lowest Observed Adverse Effect Level (LOAEL) and lowest dose tested (Fitzhugh et al., 1964). This dose was divided by a composite uncertainty factor of 1,000 (to account for rat-to-human extrapolation, potentially sensitive human subpopulations, and the use of a LOAEL rather than a No Observed Adverse Effect Level–NOAEL) to yield an oral RfD for aldrin of 3×10^{-5} mg/kg bw-day. Similarly for dieldrin, a chronic rat study NOAEL for liver toxicity of approximately 0.005 mg/kg bw-day

(Walker et al., 1969) was divided by a composite uncertainty factor of 100 (to account for rat-to-human extrapolation and potentially sensitive human subpopulations) to yield an oral RfD of 5 x 10^{-5} mg/kg bw-day.

Based upon the mouse long-term bioassays discussed in section 4.1, the USEPA has classified both aldrin and dieldrin as group B2 carcinogens under the current cancer guidelines (USEPA, 1986), i.e., as probable human carcinogens with little or no evidence of carcinogenicity in humans, and sufficient evidence in animals. Mechanistic studies performed *in vitro* and *in vivo* suggest that one or more non-genotoxic modes of action may underlie or contribute to the carcinogenic potential of aldrin and dieldrin, but these effects are not completely established, nor can a role for genotoxic mechanisms confidently be eliminated based upon the available data. Based upon these considerations, the quantitative cancer risk assessments of aldrin and dieldrin have been conducted conservatively using the linear-default model.

This approach has yielded geometric mean cancer potency estimates for aldrin and dieldrin of 17 and 16 $(mg/kg\ bw\text{-}day)^{-1}$, respectively. Using this potency, a concentration of one microgram of aldrin or dieldrin in a liter of water may be associated with a theoretical cancer risk of 4.9 deaths in a population of 10,000 for aldrin or 4.6 deaths for dieldrin, respectively. For both compounds, a drinking water concentration of 0.002 µg/L would lead to an estimated increase in lifetime cancer risk of 10^{-6}. This concentration, 0.002 µg/L, was selected as the Health Reference Level (HRL) for each chemical. The HRLs are benchmark values that are used in evaluating the occurrence data (section 3.3) while risk assessments for the contaminants are being developed.

4.3 Relative Source Contribution

Analysis of relative source contribution compares the magnitude of exposures expected via consumption of drinking water with those estimated for other relevant media such as food, air and soil. The data summarized in section 3.3 provide the basis for estimating the amounts of aldrin and dieldrin ingested via drinking water in exposed populations. A less-conservative estimate was achieved by utilizing the median and 99th percentile detected concentrations derived from only Unregulated Contaminant Monitoring (UCM) Round 2 cross-section data, which will certainly underestimate to some degree the true contribution of drinking water to the exposed population's total intake of aldrin/dieldrin.

For a 70 kg adult consuming 2 L/day of water containing aldrin at either 0.58 µg/L (median detect concentration) or 0.69 µg/L (99th percentile detect concentration), the corresponding aldrin intakes from drinking water are 1.7 x 10^{-5} and 2.0 x 10^{-5} mg/kg bw-day, respectively. For a 10 kg child consuming 1 L/day of water, the comparable values are 5.8 x 10^{-5} and 6.9 x 10^{-5} mg/kg bw-day.

Similarly, for median and 99th percentile detected concentrations of dieldrin (0.16 and 1.36 µg/L, respectively), the corresponding adult drinking water intakes of dieldrin are 0.46 x 10^{-5} and 3.9 x 10^{-5} mg/kg bw-day, respectively. Dieldrin drinking water intake values for the 10 kg child are 1.6 x 10^{-5} and 14 x 10^{-5} mg/kg bw-day.

Chapter 4.0 of the Drinking Water Support Document for Aldrin and Dieldrin (USEPA, 2001) presents data on the estimated daily dietary intake of aldrin and dieldrin. Combining estimates for non-fish food with those for fish and shellfish, adult and child dietary intakes of aldrin are estimated at 3.3-6.5 x 10^{-5} and 13-18 x 10^{-5} mg/kg bw-day, respectively. For dieldrin, the comparable adult and child dietary intakes are 3.6 x 10^{-5} and 14 x 10^{-5} mg/kg bw-day.

Comparing these derived estimates for intakes via drinking water and diet, the ratios of dietary intake to drinking water intake for aldrin range from 1.7 to 3.8 across all combinations of

age and drinking water concentration level. For dieldrin, the food/water intake ratios for adults and children are 0.9 and 1.0 using the 99th percentile water concentration, and 7.8 and 8.8 using the median water concentration. Applying the more conservative aldrin/dieldrin water concentrations based upon the monitoring data of all reporting UCM Round 2 States would reduce all of these food/water ratios by a factor of approximately 3 to 6. Thus, when conservatively analyzed relative to the diet, drinking water could potentially be responsible for a significant portion of total daily intake of aldrin/dieldrin, but only for limited populations under exposure circumstances that are considered unlikely.

The estimated daily intakes of aldrin and dieldrin from air for adults and children are small (ranging from 0.013×10^{-5} to 0.24×10^{-5}) relative to the unlikely, but potential, drinking water intakes and dietary intakes discussed above. Although soil data were not available for aldrin, data for dieldrin indicates that ingestion of soil represents only a minor exposure pathway for these compounds.

4.4 Sensitive Populations

The available literature did not provide direct evidence for any human subpopulations that are particularly sensitive to the toxic affects of aldrin/dieldrin, or for which the relevant information on absorption, metabolism, or elimination of these chemicals are known to be significantly different from those for the general population. Speculatively, fetuses and very young children might be at increased risk from exposures to aldrin/dieldrin as a result of the limited ability of the immature system to transform these contaminants in the liver to less toxic chemicals that would be eliminated from the exposed individuals; and as a result would have a higher vulnerability during these critical periods of development. Several mechanistic studies, which describe the prenatal effects of aldrin/dieldrin on GABA receptor malfunctions and on subsequent behavioral impairment, may suggest an increased sensitivity of children (Brannen et al., 1998; Liu et al., 1998; Johns et al., 1998; Castro et al., 1992). Declining organ and immune functions may also render the elderly more susceptible to aldrin/dieldrin toxicity. Additionally, it is reasonable to expect that individuals with compromised liver, immune or neurological functions (as a result of disease, genetic predisposition or other toxic insult) might also display increased sensitivity to these compounds. However, no study has convincingly demonstrated sensitivity of any one group.

4.5 Exposure and Risk Information

Because the HRL of 0.002 µg/L for these compounds is below the Minimum Reporting Level (MRL), any recorded detection will be above all three reference levels (MRL, ½ HRL, HRL). Therefore, estimates of the national population exposed to concentrations greater than any of these levels will be equivalent. For aldrin, data from only the UCM Round 2 cross-section (less-conservative approach or probable underestimate) indicate that about 39,000 people are served by public water systems (PWSs) with detections greater than the MRL, ½ HRL, and HRL. Using the more conservative data from all reporting Round 2 UCM States (a probable overestimate), approximately 1,052,000 people are served by systems with detections greater than the MRL, ½ HRL, and HRL. The corresponding estimates for dieldrin are approximately 150,000 people and 793,000 people, respectively.

For aldrin, the median and 99th percentile concentrations of detections based upon all Round 2 UCM data were 0.18 and 4.40 µg/L, respectively. Based only upon the 19-State Round 2 cross-section data, the corresponding values are 0.58 and 0.69 µg/L. The respective two sets of values for dieldrin are 0.42 and 4.40 µg/L, and 0.16 and 1.36 µg/L. While these values are above the HRL of 0.002 µg/L, it is necessary to consider that the corresponding values for all samples were below the detection limit, and that the HRL itself is likely a very conservative estimate of any human risk resulting from exposure to these chemicals.

4.6 Conclusion

In conclusion, while there is evidence that aldrin and dieldrin may have adverse health effects in humans, including the potential to cause cancer, these chemicals are detected very infrequently and generally at very low concentrations in drinking water. Furthermore, there should be no new releases to the environment because the compounds have not been used in the United States since 1987. Therefore, it is unlikely that aldrin or dieldrin will occur at frequencies or concentrations that are of public health concern or that regulation represents a meaningful opportunity for health risk reduction in persons served by public water systems. All CCL regulatory determinations and further analysis are formally presented in the *Federal Register* Notices (USEPA, 2002a; 67 FR 38222; and USEPA, 2003a; 68 FR 42898).

5.0 TECHNOLOGY ASSESSMENT

If a determination has been made to regulate a contaminant, SDWA requires development of proposed regulations within two years of making the decision. It is critical to have suitable monitoring methods and treatment technologies to support regulation development according to the schedules defined in the SDWA.

5.1 Analytical Methods

The availability of analytical methods does not influence EPA's determination of whether or not a CCL contaminant *should* be regulated. However, before EPA actually regulates a contaminant and establishes a Maximum Contaminant Level (MCL), there must be an analytical method suitable for routine monitoring. Therefore, EPA needs to have approved methods available for any CCL regulatory determination contaminant before it is regulated with an NPDWR. These methods must be suitable for compliance monitoring and should be cost effective, rapid, and easy to use.

Aldrin and dieldrin are unregulated contaminants for which monitoring was required under the Unregulated Contaminant Monitoring Program (USEPA, 1987; 52 FR 25690). Monitoring for aldrin and dieldrin was initiated through rulemaking in 1991 (USEPA, 1991b; 56 FR 3526), and began in 1993. The contaminants already have well-documented analytical methods developed specifically for low-level drinking water analyses (see Table 5-1).

5.2 Treatment Technology

Treatment technologies also do not influence the determination of whether or not a contaminant should be regulated. But before a contaminant can be regulated with an NPDWR, treatment technologies must be readily available. EPA's Office of Research and Development (ORD) has researched treatment technologies for all of the organic compounds listed as regulatory determination priorities on the CCL, including aldrin and dieldrin. The two appropriate technologies reviewed were granular activated carbon (GAC) and air stripping.

Granular activated carbon treatment removes contaminants via the physical and chemical process of sorption, by which the contaminants attach to the carbon surface as water passes through the carbon bed. Activated carbon has a large sorption capacity for many water impurities including synthetic organic contaminants, taste and odor causing compounds, and some species of mercury. Adsorption capacity is typically represented by the Freundlich isotherm constants, with higher Freundlich (K) values indicating greater sorption potential.

Air stripping involves the continuous contact of air with the water being treated, allowing volatile dissolved contaminants to transfer from the source water to the air. After contact, the

"contaminated air" is swept from the system, taking the contaminant out of contact with the treated water. The driving force for the water-to-air transfer of the volatile contaminants is the contaminant's concentration gradient between the water and air. The Henry's Law constant is a commonly used indicator of the tendency of a contaminant to partition from water to air. A larger Henry's constant indicates a greater

Table 5-1: Analytical methods for aldrin and dieldrin

Method	Type	Method Detection Limit (µg/L) for aldrin	Method Detection Limit (µg/L) for dieldrin
EPA 525.2	Gas Chromatography (GC)/Quadrupole Mass Spectrometry (MS)	0.11	0.053
	GC/Ion Trap MS	0.045	0.11
EPA 505	GC/Electron Capture Detectors (ECD)	0.075	0.012
EPA 508	GC/ECD	0.014	0.011
EPA 508.1	GC/ECD	0.009	0.003

equilibrium of the contaminant in the air. Thus, contaminants having larger Henry's constants are more easily removed by air stripping.

Predictive computer modeling and specific chemical characteristics were used to determine the isotherm constants needed to evaluate the two treatment technologies. The rule of thumb used for SDWA compounds, learned through the development of cost-and-technology documents to support other drinking water regulations, is that GAC is considered to be cost-effective if the contaminant has a Freundlich (K) value above 200 (Speth and Adams, 1993). For air stripping, a compound with a Henry's constant above dibromochloropropane (0.005) or ethylene dibromide (0.037) is considered strippable at a reasonable cost.

Since aldrin has a predicted Freundlich (K) value of 718,000, it can be effectively treated by the GAC method. However, because its predicted Henry's Law constant is 0.027, aldrin can only undergo effective air stripping procedures under certain concentration conditions. Dieldrin has a predicted Freundlich (K) value of 486,000 and a predicted Henry's Law constant of 2.0×10^{-5}. Therefore, only GAC is an applicable treatment technology for dieldrin. Its low volatilization potential makes air stripping impractical.

6.0 SUMMARY AND CONCLUSIONS - DETERMINATION OUTCOME

Three statutory criteria are used to guide the determination of whether regulation of a CCL contaminant is warranted: 1) the contaminant may adversely affect the health of persons; 2) the contaminant is known or is likely to occur in public water systems with a frequency, and at levels, of public health concern; and 3) regulation of the contaminant presents a meaningful opportunity for health risk reduction for persons served by public water systems. As required by SDWA, a decision to regulate a contaminant commits the EPA to propose a Maximum Contaminant Level Goal (MCLG) and promulgate an NPDWR for the contaminant. A decision not to regulate a contaminant is considered a final Agency action and is subject to judicial review. The Agency can choose to publish a Health Advisory (a nonregulatory action) or other guidance for any contaminant on the CCL that does not meet the criteria for regulation.

The available toxicological data indicate that aldrin and dieldrin have the potential to cause adverse health effects in humans and animals. In humans, the most common manifestation of acute aldrin/dieldrin toxicity is neurotoxicity to the central nervous system, including hyperirritability, convulsions, and coma, in some cases followed by cardiovascular effects such as unusually rapid beating of the heart and elevated blood pressure. In smaller doses over time, aldrin and dieldrin may cause headache, dizziness, general malaise, nausea, vomiting, and muscle twitching or muscle spasms. Aldrin and dieldrin toxicity may also be responsible for reported cases of immunohemolytic and aplastic anemia. Indications of aldrin/dieldrin toxicity in animal subjects include neurotoxicity, hepatotoxicity (e.g., elevated serum enzyme levels, unregulated liver cell regenerations, rapid turn-over in bile duct cells, localized cell degeneration and death), changes in the tissue structure of the liver, renal lesions and exacerbation of pre-existing kidney illness. Aldrin and dieldrin appear to be weak endocrine disruptors, affecting the reproductive system of animal subjects. Though their carcinogenicity in mice is established, no occupational or epidemiological study has convincingly linked aldrin and dieldrin to cancer in humans.

Aldrin and dieldrin are synthetic organic compounds (SOCs) that have been used as agricultural insecticides (primarily on corn and citrus products) and termite deterrents. Production and most agricultural uses of aldrin and dieldrin ceased in the U.S. in 1974, and all uses and imports were discontinued in 1987. The presence and persistence of aldrin and dieldrin in the environment is evidenced by detections of the compounds in hazardous waste sites in at least 31 and 38 States, respectively (at NPL sites), as well as detections of both chemicals in site samples in at least 40 States (listed in ATSDR's HazDat).

In the ambient environment, aldrin and dieldrin are occasionally detected at low concentrations. Both substances are hydrophobic, and thus are more likely to sorb to surface soil or enter biotic tissue than enter groundwater aquifers. Aldrin has been detected more commonly in streambed samples in agricultural land use areas than in other land use areas or in biotic tissue. Dieldrin has been found to occur most commonly in agricultural surface waters and shallow urban aquifers; it has also been detected in biotic tissue, suggesting a risk of bioaccumulation.

Monitoring data indicate that aldrin and dieldrin are infrequently detected in public water supplies, with only 0.006% and 0.06% of all cross-section samples showing detections for aldrin and dieldrin, respectively. Significantly, the values for the 99th percentile and median concentrations of all cross-section samples are less than the Minimum Reporting Level (MRL) for both contaminants. According to the cross-section model, less than 0.02% of the national population served by public water supplies (40,000-50,000 people) is served by PWSs with aldrin detections and less than 0.1% of the population (150,000 people) is served by PWSs with dieldrin detections. Using more conservative estimates of occurrence from all States reporting SDWA Round 2 monitoring data, including States with biased data, those figures increase to

only 0.5% of the PWS population served (approximately 1,052,000 people) for aldrin, and 0.4% of the PWS population served (approximately 793,000 people) for dieldrin. (For details of the methodology used to construct the national cross-section, see section 3.)

Additional data from corn belt States, where use of aldrin and dieldrin was historically high, were evaluated to supplement the cross-section data. Drinking water data from Iowa, Indiana, and Illinois also show low occurrence of aldrin and dieldrin. There were no detections in either surface or ground water PWSs in the State of Iowa. Illinois and Indiana reported no groundwater detections; surface water detections were low, with 1.8% of Illinois' surface water systems (0.1% of samples) showing detections of aldrin/dieldrin and 2.1% of Indiana's surface water systems (0.3% of samples) showing detections of dieldrin alone. For Illinois and Indiana surface water PWSs, the 99th percentile concentrations of all samples were below the reporting level and the maximum concentrations were 0.1 µg/L and 0.04 µg/L, respectively. Moreover, in a survey of Illinois community water supply wells dieldrin was detected in only 1.6% of all sampled wells.

EPA considers exposure to both the general public and sensitive populations, including fetuses, infants, and children, in making its regulatory determination. A factor of ten is included in the reference dose to account for differences in sensitivity between human subpopulations. Besides drinking water, food intake may also present a risk of exposure to aldrin and dieldrin, of a similar order of magnitude, to both children and adults. (Exposure via air and soil are considered less significant.)

In conclusion, while there is evidence that aldrin and dieldrin have adverse health effects in humans, their occurrence in drinking water at frequencies or concentrations significant for public health concern is low. Furthermore, occurrence of aldrin and dieldrin in drinking water supplies in the coming years is likely to decrease, since the substances are no longer produced or used commercially. Therefore regulation of aldrin and dieldrin may be unlikely to represent a meaningful opportunity for health risk reduction. All CCL regulatory determinations and further analysis are formally presented in the *Federal Register* Notices (USEPA, 2002a; 67 FR 38222; and USEPA, 2003a; 68 FR 42898).

This page intentionally left blank.

REFERENCES

Amaoteng-Adjepong, Y., N. Sathiakumar, E. Delzell and P. Cole. 1995. Mortality Among Workers at a Pesticide Manufacturing Plant. *J. Occup. Environ. Med.* 37:471-478 (as cited in Stevenson *et al.*, 1999).

American Conference of Governmental Industrial Hygienists (ACGIH). 1984. *Documentation of the Threshold Limit Values for Substances in Workroom Air.* Third Edition. Cincinnati, OH: ACGIH. 139 pp. (as cited in USEPA, 1988).

Agency for Toxic Substances and Disease Registry (ATSDR). 1993. *Toxicological Profile for Aldrin/Dieldrin (update)*. Atlanta: Agency for Toxic Substances and Disease Registry. 184 pp.

ATSDR. 2000. *Toxicological Profile for Aldrin/Dieldrin (Update)*. Atlanta, GA: Agency for Toxic Substances and Disease Registry. 280 pp.

Black A.M.S. 1974. Self-Poisoning with Dieldrin: A Case Report and Pharmacokinetic Discussion. *Anesth. Intensive Care* 2:369-374 (as cited in ATSDR, 2000).

Borgmann, A.R., C.H. Kitselman, P.A. Dahm, J.E. Pankaskie and F.R. Dutra. 1952. Toxicological Studies of Dieldrin on Small Laboratory Animals. Unpublished report by Kansas State College (as cited in USEPA, 1992).

Brannen, K.C., L.L. Devaud, J. Liu, and J.M. Lauder. 1998. Prenatal Exposure to Neurotoxicants Dieldrin or Lindane Alters *Tert*-Butylbicyclophosphorothionate Binding to GABA(A) Receptors in Fetal Rat Brainstem. *Dev. Neurosci.* 20:34-41.

Brown, D.P. 1992. Mortality of Workers Employed at Organochlorine Pesticide Manufacturing Plants - An Update. *Scand. J. Environ. Health* 18:155-161 (as cited in Stevenson *et al.*, 1999).

Cadmus Group, Inc (Cadmus). 2000. *Methods for Estimating Contaminant Occurrence and Exposure in Public Drinking Water Systems in Support of CCL Determinations*. Draft report to USEPA, Washington, D.C., by Cadmus Group, Waltham, MA, July 25, 2000.

Casteel, S.W., F.T. Satalowich, J.D. Kendall, G.E. Rottinghaus, H.S. Gosser, and N.R. Schneider. 1993. Aldrin Intoxication and Clearance of Associated Dieldrin Residues in a Group of Feedlot Cattle. *J. Am. Vet. Med. Assoc.* 202:83-85.

Castro,V.L., M.M. Bernardi, and J. Palermo-Neto. 1992. Evaluation of Prenatal Aldrin Intoxication in Rats. *Arch. Toxicol.* 66:149-152.

Davis, K.J. 1965. Pathology Report on Mice Fed Aldrin, Dieldrin, Heptachlor or Heptachlor Epoxide for Two Years. Internal FDA memorandum to Dr. A.J. Lehman. FDA. July 19. (as cited in USEPA, 1987, 1992).

Davis, K.J. and O.G. Fitzhugh. 1962. Tumorigenic Potential of Aldrin and Dieldrin for Mice. *Toxicol. Appl. Pharmacol.* 4:187-189 (as cited in USEPA, 1987, 1992).

de Jong, G., G.M. Swaen and J.J. Slangen. 1997. Mortality of Workers Exposed to Dieldrin and Aldrin: a Retrospective Cohort Study. *Occup. Environ. Med.* 54:702-707.

de Jong, G. 1991. Long-Term Health Effects of Aldrin and Dieldrin: A Study of Exposure, Health Effects and Mortality of Workers Engaged in the Manufacture and Formulation of the Insecticides Aldrin and Dieldrin. *Toxicol. Lett. Suppl.* 1-206 (as cited in Stevenson *et al.*, 1999).

Dean, B.J., S.M.A. Doak and H. Somerville. 1975. The Potential Mutagenicity of Dieldrin (HOED) in Mammals. *Food Cosmet. Toxicol.* 13:317-323 (as cited in ATSDR, 2000).

Deichmann, W.B., W.E. MacDonald, E. Blum, M. Bevilacqua, J. Radomski, M. Keplinger and M. Balkus. 1970. Tumorigenicity of Aldrin, Dieldrin and Endrin in the Albino Rat. *Ind. Med. Surg.* 39(10):426-434 (as cited in USEPA, 1987, 1992).

Deichmann, W.B. 1974. Statement of Testimony for Aldrin/Dieldrin Hearings Suspension Phase, before the U.S. Environmental Protection Agency. August 23, 1974 (as cited in USEPA, 1987).

Deichmann, W.B., M. Keplinger, F. Sala, and E. Glass. 1967. Synergism Among Oral Carcinogens in the Simultaneous Feeding of Four Tumorigens to Rats. *Toxicol. Appl. Pharmacol.* 11:88-103 (as cited in USEPA, 1987).

Ditraglia, D., D.P. Brown, T. Namekata, and N. Iverson. 1981. Mortality Study of Workers Employed at Organochlorine Pesticide Manufacturing Plants. *Scand. J. Work. Environ. Health.* 4:140-146 (as cited in Stevenson *et al.*, 1999).

Epstein, S.S., E. Arnold, J. Andrea, W. Bass, and Y. Bishop. 1972. Detection of Chemical Mutagens by the Dominant Lethal Assay in the Mouse. *Toxicol Appl. Pharmacol* 23:288-325 (as cited in ATSDR, 2000).

Fitzhugh, O.G., A.A. Nelson, and M.L. Quaife. 1964. Chronic Oral Toxicity of Aldrin and Dieldrin in Rats and Dogs. *Food Cosmet. Toxicol.* 2:551-562.

Genetic Activity Profiles. 2000. Data record for aldrin. Database and software are a joint effort of the U.S. Environmental Protection Agency and the International Agency for Research on Cancer. Lohman, P.H.M. and W.J.A. Lohman, authors. Downloaded from http://www.epa.gov/gapdb on January 19, 2001.

Goetsch, W.D., D.P. McKenna, and T.J. Bicki. 1992. *Statewide Survey for AgriculturalCchemicals in Rural, Private Water-Supply Wells in Illinois*. Springfield, IL: Illinois Department of Agriculture, Bureau of Environmental Programs. 4 pp.

Good, E.E. and G.W. Ware. 1969. Effects of Insecticides on Reproduction in the Laboratory Mouse. IV. Endrin and Dieldrin. *Toxicol Appl Pharmacol*. 14:201-203 (as cited in ATSDR, 2000).

Goolsby, D.A. and W.A. Battaglin. 1993. Occurrence, Distribution and Transport of Agricultural Chemicals in Surface Waters of the Midwestern United States. In: *Selected Papers on Agricultural Chemicals in Water Resources of the Midcontinental United States*, Ed. D.A. Goolsby, L.L. Boyer, and G.E. Mallard. U.S. Geological Survey Open-File Report 94-418. pp. 1-25.

Hallberg, G.R., D.G. Riley, J.R. Kantamneni, P. J. Weyer, and R.D. Kelley. 1996. *Assessment of Iowa Safe Drinking Water Act Monitoring Data: 1988-1995*. Research Report No. 97-1. Iowa City: The University of Iowa Hygienic Laboratory. 132 pp.

Hamilton, H.E., D.P. Morgan, and A. Simmons. 1978. A Pesticide (Dieldrin)-Induced Immunohemolytic Anemia. *Environ. Res.* 17:155-164 (as cited in USEPA, 1992).

Harr J.R., R.R. Claeys, J.F. Bone, and T.W. McCorde. 1970. Dieldrin Toxicosis: Rat Reproduction. *Am. J. Vet. Res.* 31:181-189 (as cited in ATSDR, 2000).

Howard, Philip H. 1991. *Handbook of Environmental Fate and Exposure Data for Organic Chemicals. Volume III: Pesticides.* Chelsea, MI: Lewis Publishers, Inc. 684 pp.

Jager, K.W. 1970. *Aldrin, Dieldrin, Endrin and Telodrin: An Epidemiological and Toxicological Study of Long-Term Occupational Exposure.* New York: Elsevier Publishing Company. 234 pp. (as cited in USEPA, 1980).

Johns, J.M., J. Liu, N. Bhasin, D.R. Grayson, L.L. Devaud, S. Moy, D. Lubin, and J.M. Lauder. 1998. Prenatal Exposure to Organochlorine Pesticide Dieldrin: Effects of Postnatal Expression of $GABA_A$ Receptors and Behavior in the Rat. *Soc. of Neur.* 24(Pt 1):101.

Klaassen, C.D. 1996. *The Basic Science of Poisons: Toxic Effects of Pesticides.* 5th ed. Ed. C.D. Klaassen, M.O. Amdur, J. Doull. New York: McGraw-Hill. pp. 652-653.

Kolpin, D.W., J.E. Barbash, and R.J. Gilliom. 1998. Occurrence of Pesticides in Shallow Groundwater of the United States: Initial Results from the National Water Quality Assessment Program. *Env. Sci. Tech*. 32:558-566.

Kolpin, D.W., J.E. Barbash, and R.J. Gilliom. 2000. Pesticides in Ground Water of the United States, 1992-1996. *Ground Water*. 38(6):858-863.

Krzystyniak, K., P. Hugo, D. Flipo, and M. Fournier. 1985. Increased Susceptibility to Mouse Hepatitis Virus 3 of Peritoneal Macrophages Exposed to Dieldrin. *Toxicol. Appl. Pharmacol.* 80:397-408 (as cited in ATSDR, 2000).

Kurata, M., K. Hirose, and M. Umeda. 1982. Inhibition of Metabolic Cooperation in Chinese Hamster Cells by Organochlorine Pesticides. *Jpn. J. Cancer Res.* 73:217-221 (as cited in Genetic Activity Profiles. 2000).

Larson, S.J., R.J. Gilliom, and P.D. Capel. 1999. *Pesticides in Streams of the United States--Initial Results from the National Water-Quality Assessment Program.* U.S. Geological Survey Water-Resources Investigations Report 98-4222. 92 pp. Available on the Internet at: URL: http://water.wr.usgs.gov/pnsp/rep/wrir984222/

Leahy, P.P. and T.H. Thompson. 1994. *The National Water-Quality Assessment Program*. U.S. Geological Survey Open-File Report 94-70. 4 pp. Available on the Internet at: http://water.usgs.gov/nawqa/NAWQA.OFR94-70.html. Last modified August 23, 2000.

Liu, J., K.C. Brannen, D.R. Grayson, A.L. Morrow, L.L. Devaud, and J.M. Lauder. 1998. Prenatal Exposure to the Pesticide Dieldrin or the $GABA_A$ Receptor Antagonist Bicuculline Differentially Alters Expression of $GABA_A$ Receptor Subunit mRNAs in Fetal Rat Brainstem. *Dev. Neurosci.* 20:83-92.

Liu, J., A.L. Morrow, L.L. Devaud, D.R. Grayson, and J.M. Lauder. 1997. Regulation of $GABA_A$ Receptor Subunit mRNA Expression by the Pesticide Dieldrin in Embryonic Brainstem Cultures: A Quantitative Competitive Reverse Transcription-Polymerase Chain Reaction Study. *J. Neurosci. Res.* 49:645-653.

Loose L.D., J.B. Silkworth , T. Charbonneau, and F. Blumenstock. 1981. Environmental Chemical Induced Macrophage Dysfunction. *Environ. Health Perspect.* 39:79-92 (as cited in ATSDR, 2000).

Lu, F.C., D.C. Jessup, and A. Lavallee. 1965. Toxicity of Pesticide in Young Versus Adult Rats. *Food Cosmet. Toxicol.* 3:591-596 (as cited in ATSDR, 2000).

Loose, L.D. 1982. Macrophage Induction of T-Suppressor Cells in Pesticide-Exposed and Protozoan-Infected Mice. *Environ Health Perspect.* 43:89-97 (as cited in ATSDR, 2000).

MacDonald, W.E., W.A.D. Anderson, M. Bevilacqua, E. Blum, and W.B. Deichmann. 1972. The Tumorigenicity of Dieldrin in the Swiss-Webster Mouse. Unpublished report (as cited in USEPA, 1987).

Meierhenry, E.F., B.H. Reuber, M.E. Gershwin, L.S. Hsieh, and S.W. French. 1983. Dieldrin-Induced Mallory Bodies in Hepatic Tumors of Mice of Different Strains. *Hepatology* 3:90-95 (as cited in USEPA, 1987).

Mikalsen, S.O. and T. Sanner. 1993. Intercellular Communication in Colonies of Syrian Hamster Embryo Cells and the Susceptibility for Morphological Transformation. *Carcinogenesis* 14:251-257 (as cited in Genetic Activity Profiles. 2000).

Miller, T. 2000. *Selected Findings and Current Perspectives on Urban Water Quality-The National Water Quality Assessment (NAWQA) Program of the U.S. Geological Survey*. Paper presented to the NAWQA National Liaison Committee, June 13, 2000. 8 pp.

Miller, T.L. and W. G. Wilber. 1999. Emerging Drinking Water Contaminants: Overview and Role of the National Water Quality Assessment Program. In: *Identifying Future Drinking Water Contaminants*. Washington, D.C.: National Academy Press.

Muirhead, E. E, M. Groves, R. Guy et al. 1959. Acquired Hemolytic Anemia, Exposure to Insecticides and Positive Coombs Test Dependent on Insecticide Preparations. *Vox Sang* 4:277-292 (as cited in ATSDR, 2000).

Nagata, K. and T. Narahashi. 1994. Dual Action of the Cyclodiene Insecticide Dieldrin on the Gamma-Aminobutyric Acid Receptor-Chloride Channel Complex of Rat Dorsal Root Ganglion Neurons. *J. Pharmacol. Exp. Ther.* 269:164-171.

Nagata, K., B.J. Hamilton, D.B. Carter, and T. Narahashi. 1994. Selective Effects of Dieldrin on the $GABA_A$ Receptor-Channel Subunits Expressed in Human Embryonic Kidney Cells. *Brain Res.* 645:19-26.

Nagata, K. and T. Narahashi. 1995. Multiple Actions of Dieldrin and Lindane on the $GABA_A$ Receptor-Chloride Channel Complex of Rat Dorsal Root Ganglion Neurons. *Pestic. Sci*. 44:1-7.

National Cancer Institute (NCI). 1978. *Bioassays of Aldrin and Dieldrin for Possible Carcinogenicity.* National Cancer Institute Carcinogenesis Technical Report Series, NCI-CG-TR-21 and NCI-CG-TR-22 (as cited in ATSDR, 2000).

Nowell, L. 1999. *National Summary of Organochlorine Detections in Bed Sediment and Tissues for the 1991 NAWQA Study Units*. Available on the Internet at: http://water.wr.usgs.gov/pnsp/rep/bst/. Last updated October 18, 1999.

Pick A., H. Joshua, M. Leffkowitz et al. 1965. [Aplastic Anemia Following Exposure to Aldrin.] *Medicine* 68:164-167 (Hebrew; as cited in ATSDR, 2000).

Sanchez-Ramos, J., A. Facca, A. Basit, and S. Song. 1998. Toxicity of Dieldrin for Dopaminergic Neurons in Mesencephalic Cultures. *Exp. Neurol.* 150:263-271.

Song, J. and W.E. Harville. 1964. Carcinogenicity of Aldrin and Dieldrin in Mouse and Rat Liver. *Fed. Proc. Fed. Am. Soc. Exp. Biol.* 23:336 (as cited in USEPA, 1987).

Speth, T.F. and J.Q. Adams. 1993. GAC and Air-Stripping Design Support for the Safe Drinking Water Act. In: *Strategies and Technologies for Meeting SDWA Requirements*. Eds. R. Clark and R.S. Summers. Lancaster, PA: Technomic Publ. Co.

Spiotta, E.J. 1951. Aldrin Poisoning in Man. *Arch. Ind. Hyg. Occup. Med.* 4:560-566 (as cited in USEPA, 1992).

Stevenson, D.E., E.F. Walborg, Jr., D.W. North, R.L. Sielken, Jr., C.E. Ross, A.S. Wright, Y. Xu, L.M. Kamendulis, and J.E. Klaunig. 1999. Monograph: Reassessment of Human Cancer Risk of Aldrin/Dieldrin. *Toxicol. Lett.* 109:123-186.

Tennekes, H.A., A.S. Wright, and K.M. Dix. 1979. The Effects of Dieldrin, Diet, and Other Environmental Components on Enzyme Function and Tumor Incidence in Livers of CF_1 Mice. *Arch. Toxicol.* 2:197-212 (as cited in USEPA, 1987).

Thorpe, E. and A.I.T. Walker. 1973. The Toxicology of Dieldrin (HEOD). Part II. Comparative Long-Term Oral Toxicology Studies in Mice with Dieldrin, DDT, Phenobarbitone, beta-BHC and gamma-BHC. *Food Cosmet. Toxicol.* 11:433-441 (as cited in USEPA, 1987).

Treon, J.F. and F.P. Cleveland. 1955. Toxicity of Certain Chlorinated Hydrocarbon Insecticides for Laboratory Animals, with Special Reference to Aldrin and Dieldrin. *Agric. Food Chem.* 3: 402-408 (as cited in USEPA, 1992).

US Environmental Protection Agency (USEPA). 1980. *Ambient Water Quality Criteria for Aldrin/Dieldrin.* EPA Report 440-5-80-019. 150 pp.

USEPA. 1986. Guidelines for Carcinogen Risk Assessment. *Federal Register* 51, no. 185 (24 Sept): 33992.

USEPA. 1987. National Primary Drinking Water Regulations-Synthetic Organic Chemicals; Monitoring for Unregulated Contaminants; Final Rule. *Federal Register* 52, no. 130 (8 July): 25690.

USEPA. 1988. *Health Advisories for 50 Pesticides (Including...Dieldrin...)*. Washington, D.C.: Office of Drinking Water. 861 pp.

USEPA. 1991a. *Drinking Water Health Advisory for Aldrin*. Washington, D.C.: Office of the Assistant Administrator for Water. 27 pp.

USEPA. 1991b. National Primary Drinking Water Regulations - Synthetic Organic Chemicals and Inorganic Chemicals; Monitoring for Unregulated Contaminants; National Primary Drinking Water Regulations Implementation; National Secondary Drinking Water Regulations; Final Rule. *Federal Register* 56, no. 20 (30 January): 3526.

USEPA. 1992. Drinking Water; National Primary Drinking Water Regulations – Synthetic Organic Chemicals and Inorganic Chemicals; National Primary Drinking Water Regulations Implementation. *Federal Register* 57, no. 138 (17 July): 31776.

USEPA. 1996. *Emergency Planning and Community Right-to-Know Section 313, List of Toxic Chemicals*. Washington, D.C.: USEPA. Available on the Internet at: http://www.epa.gov/tri/chemls2.pdf Last modified March 23, 2000. Link to site at: http://www.epa.gov/tri/chemical.htm

USEPA. 1998a. Announcement of the Drinking Water Contaminant Candidate List; Notice. *Federal Register* 63, no. 40 (2 March): 10274.

USEPA. 1999a. Suspension of Unregulated Contaminant Monitoring Requirements for Small Public Water Systems; Final Rule and Proposed Rule. *Federal Register* 64, no. 5 (8 January): 1494.

USEPA. 1999b. Revisions to the Unregulated Contaminant Monitoring Regulation for Public Water Systems; Final Rule. *Federal Register* 64, no. 180 (17 September): 50556.

USEPA. 1999c. *Superfund Hazardous Waste Site Basic Query Form*. Washington, D.C.: USEPA. Available on the Internet at: http://www.epa.gov/superfund/sites/query/basic.htm Last modified December 1,1999.

USEPA. 1999d. *A Review of Contaminant Occurrence in Public Water Systems*. Office of Water. EPA Report 816-R-99-006. 78 pp.

USEPA. 2000a. *TRI Explorer: Geographic Report*. Washington, D.C.: USEPA. Available on the Internet at: http://www.epa.gov/triexplorer/geography.htm Last modified May 5, 2000.

USEPA. 2000b. *The Toxic Release Inventory (TRI) and Factors to Consider When Using TRI Data*. Washington, D.C.: USEPA. Available on the Internet at: http://www.epa.gov/tri/tri98/98over.pdf Last modified August 11, 2000. Link to site at: http://www.epa.gov/tri/tri98.

USEPA. 2000c. *What is the Toxic Release Inventory?* Washington, D.C.: USEPA. Available on the Internet at: http://www.epa.gov/tri/general.htm Last modified February 28, 2000.

USEPA. 2000d. *Water Industry Baseline Handbook*. Second Edition (Draft). Washington, D.C: USEPA.

USEPA. 2000e. *Regulatory Matrix of TRI Chemicals in Other Federal Programs.* Washington, D.C.: USEPA. Available on the Internet at: www.epa.gov/tri/chemicals.htm Last modified 3/23/00.

USEPA. 2001a. *Analysis of National Occurrence of the 1998 Contaminant Candidate List Regulatory Determination Priority Contaminants in Public Water Systems*. Office of Water. EPA report 815-D-01-002. 77 pp.

USEPA. 2001b. *Occurrence of Unregulated Contaminants in Public Water Systems: An Initial Assessment.* Office of Water. EPA report 815-P-00-001. Office of Water. 50 pp.

USEPA. 2002a. Announcement of Preliminary Regulatory Determinations for Priority Contaminants on the Drinking Water Contaminant Candidate List. *Federal Register* 67, no. 106 (3 June): 38222.

USEPA. 2002b. *Occurrence Estimation Methodology and Occurrence Findings Report for the Six-Year Review of Existing National Primary Drinking Water Regulations*. EPA Report 815-R-03-006, Office of Water. June.

USEPA. 2003a. Announcement of Regulatory Determinations for Priority Contaminants on the Drinking Water Contaminant Candidate List; Notice. *Federal Register* 68, no. 138 (18 July): 42898.

USEPA. 2003b. *Health Effects Support Document for Aldrin/Dieldrin*. Office of Water. EPA Report 822-R-03-001. February 2003. 256 pp.

US Geological Survey (USGS). 2000. *Pesticides in Stream Sediment and Aquatic Biota*. USGS Fact Sheet FS-092-00. 4 pp.

USGS. 1999a. *The Quality of our Nation's Waters: Nutrients and Pesticides*. USGS Circular 1225. 82 pp.

USGS. 1998. *Pesticides in Surface and Ground Water of the United States: Summary of Results of the National Water Quality Assessment Program (NAWQA): Provisional Data–Subject to Revision.* Reston, VA: USGS. Available on the Internet at: http://water.wr.usgs.gov/pnsp/allsum/ Last modified October 9, 1998.

USGS. 1999b. *Pesticides Analyzed in NAWQA Samples: Use, Chemical Analyses, and Water-Quality Criteria. Provisional Data–Subject to Revision.* Reston, VA: USGS. Available on the Internet at: http://water.wr.usgs.gov/pnsp/anstrat/ Last modified August 20, 1999.

van Raalte, H.G.S. 1977. Human Experience with Dieldrin in Perspective. *Ecotox. Environ. Saf.* 1:203-210 (as cited in ATSDR, 2000).

Versteeg, J.P.J. and K.W. Jager. 1973. Long-Term Occupational Exposure to the Insecticides Aldrin, Dieldrin, Endrin, and Telodrin. *Br. J. Ind. Med.* 30:201-202 (as cited in ATSDR, 2000).

Virgo, B.B. and G.D. Bellward. 1975. Effects of Dietary Dieldrin on Reproduction in the Swiss-Vancouver (SWV) Mouse. *Environ. Physiol. Biochem.* 5:440-450 (as cited in ATSDR, 2000).

Wade, M. H., J.E. Trosko, and M. Schindler. 1986. A Flourescence Photobleaching Assay of Gap Junction-Mediated Communication Between Human Cells. *Science* 232:525-528 (as cited in Genetic Activity Profiles. 2000).

Wagner, S.R. and F.E. Greene. 1978. Dieldrin-Induced Alterations in Biogenic Amine Content of Rat Brain. *Toxicol. Appl. Pharmacol.* 43:45-55 (as cited in ATSDR, 2000).

Walker, A.I.T., D.E. Stevenson, J. Robinson, E. Thorpe, and M. Roberts. 1969. The Toxicology and Pharmacodynamics of Dieldrin (HEOD): Two-Year Oral Exposures of Rats and Dogs. *Toxicol. Appl. Pharmacol*. 15:345-373 (as cited in ATSDR, 2000).

Walker, A.I.T., E. Thorpe, and D.E. Stevenson. 1972. The Toxicology of Dieldrin (HEOD). I. Long-Term Oral Toxicity Studies in Mice. *Food Cosmet. Toxicol.* 11:415-432 (as cited in USEPA, 1987, 1988, and 1993b).

Woolley, D., L. Zimmer, D. Dodge, and K. Swanson. 1985. Effects of Lindane-Type Insecticides in Mammals: Unsolved Problems. *Neurotoxicity* 6:165-192 (as cited in ATSDR, 2000).

Zhong-Xiang, L., T. Kavanagh, J.E. Trosko, and C.C. Chang. 1986. Inhibition of Gap Junctional Intercellular Communication in Human Teratocarcinoma Cells by Organochlorine Pesticides. *Toxicol. Appl. Pharmacol.* 83: 10-19 (as cited in Genetic Activity Profiles. 2000).

APPENDIX A: Abbreviations and Acronyms

ACGIH	- American Conference of Governmental Industrial Hygienists
ATSDR	- Agency for Toxic Substances and Disease Registry
CAS	- Chemical Abstract Service
CCL	- Contaminant Candidate List
CERCLA	- Comprehensive Environmental Response, Compensation & Liability Act
CMR	- Chemical Monitoring Reform
CWS	- community water system
DBCP	- dibromochloropropane
DNA	- deoxyribonucleic acid
ECD	- electron capture detectors
EHS	- extremely hazardous substance
EPA	- Environmental Protection Agency
EPCRA	- Emergency Planning and Community Right-to-Know Act
FR	- federal register
GABA	- gamma aminobutyric acid
GAC	- granular activated carbon (treatment technology for organic compounds)
GC	- gas chromatography (a laboratory method)
g/mol	- grams per mole
GW	- ground water
HazDat	- Hazardous Substance Release and Health Effects Database
HEOD	- 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4α,5,6,7,8,8α-octahydro-1,4-endo,exo-5,8-dimethanonaphthalene
HHDN	- 1,2,3,4,10,10-hexachloro-1,4,4α,5,8,8α-hexahydro-1,4-endo,exo-5,8-dimethanonaphthalene
HRL	- Health Reference Level
IOC	- inorganic compound
K_{oc}	- organic carbon partition coefficient
K_{ow}	- octanol-water partitioning coefficient
LD_{50}	- 50% lethal dose
LOAEL	- lowest observed adverse effect level
MCL	- maximum contaminant level
MCLG	- maximum contaminant level goal
MDL	- method detection limit
mg	- milligrams
MRL	- minimum reporting level
MS	- mass spectrometry (a laboratory method)
NAWQA	- National Water Quality Assessment Program
NCI	- National Cancer Institute
NCOD	- National Drinking Water Contaminant Occurrence Database
NDWAC	- National Drinking Water Advisory Council
NIOSH	- National Institute for Occupational Safety and Health
NIRS	- National Inorganic and Radionuclide Survey
nm	- nanometer
NOAEL	- no observed adverse effect level
NPDWR	- National Primary Drinking Water Regulation
NPL	- National Priorities List
NTNCWS	- non-transient non-community water system
OGWDW	- Office of Ground Water and Drinking Water
OMB	- Office of Management and Budget
ORD	- Office of Research and Development
OSHA	- Occupational Safety and Health Administration
PWS	- public water system

RCRA	- Resource Conservation and Recovery Act
RfD	- reference dose
SARA Title III	- Superfund Amendments and Reauthorization Act
SDWA	- Safe Drinking Water Act
SDWIS	- Safe Drinking Water Information System
SDWIS/FED	- Federal Safe Drinking Water Information System
SOC	- synthetic organic compound
SW	- surface water
TPQ	- threshold planning quantity
TRI	- Toxic Release Inventory
UCM	- Unregulated Contaminant Monitoring
UCMR	- Unregulated Contaminant Monitoring Regulation/Rule
URCIS	- Unregulated Contaminant Monitoring Information System
USDA	- United States Department of Agriculture
USEPA	- United States Environmental Protection Agency
USGS	- United States Geological Survey
VOC	- volatile organic compound
µg	- micrograms
>MCL	- percentage of systems with exceedances
>MRL	- percentage of systems with detections