



TOXICOLOGICAL REVIEW

OF

CHLORDECONE (KEPONE)

(CAS No. 143-50-0)

**In Support of Summary Information on the
Integrated Risk Information System (IRIS)**

September 2009

U.S. Environmental Protection Agency
Washington, DC

DISCLAIMER

This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

CONTENTS —TOXICOLOGICAL REVIEW OF CHLORDECONE (CAS No. 143-50-0)

| | |
|---|------|
| LIST OF TABLES | v |
| LIST OF FIGURES | viii |
| LIST OF ABBREVIATIONS AND ACRONYMS | ix |
| FOREWORD | x |
| AUTHORS, CONTRIBUTORS, AND REVIEWERS | xi |
| 1. INTRODUCTION | 1 |
| 2. CHEMICAL AND PHYSICAL INFORMATION RELEVANT TO ASSESSMENTS | 3 |
| 3. TOXICOKINETICS | 5 |
| 3.1. ABSORPTION | 5 |
| 3.2. DISTRIBUTION..... | 7 |
| 3.3. METABOLISM | 10 |
| 3.4. ELIMINATION | 13 |
| 3.5. PHYSIOLOGICALLY BASED TOXICOKINETIC MODELS..... | 15 |
| 4. HAZARD IDENTIFICATION..... | 18 |
| 4.1. STUDIES IN HUMANS—EPIDEMIOLOGY, CASE REPORTS, CLINICAL CONTROLS | 18 |
| 4.2. SUBCHRONIC AND CHRONIC STUDIES AND CANCER BIOASSAYS IN ANIMALS—ORAL AND INHALATION..... | 19 |
| 4.2.1. Subchronic Studies..... | 20 |
| 4.2.1.1. <i>Oral Exposure Studies</i> | 20 |
| 4.2.1.2. <i>Inhalation Exposure Studies</i> | 20 |
| 4.2.2. Chronic Studies..... | 20 |
| 4.2.2.1. <i>Oral Exposure Studies</i> | 21 |
| 4.3. REPRODUCTIVE/DEVELOPMENTAL STUDIES..... | 34 |
| 4.3.1. Reproductive Toxicity Studies..... | 34 |
| 4.3.2. Developmental Toxicity Studies..... | 46 |
| 4.3.3. Screening Studies..... | 48 |
| 4.4. OTHER DURATION-OR ENDPOINT-SPECIFIC STUDIES..... | 49 |
| 4.4.1. Acute Toxicity Studies..... | 49 |
| 4.4.2. Potentiation of Halomethane Toxicity | 49 |
| 4.4.3. Neurotoxicity Studies..... | 51 |
| 4.4.4. Endocrine Disruption Studies | 51 |
| 4.4.5. Immunological Studies | 53 |
| 4.5. MECHANISTIC DATA AND OTHER STUDIES IN SUPPORT OF THE MODE OF ACTION | 57 |
| 4.5.1. Genotoxicity..... | 57 |
| 4.5.2. Tumor Promotion and Mechanistic Studies..... | 57 |
| 4.5.3. Structural Analog Data—Relationship to Mirex | 60 |
| 4.6. SYNTHESIS OF MAJOR NONCANCER EFFECTS..... | 63 |
| 4.6.1. Oral | 66 |
| 4.6.2. Mode-of-Action Information—Glomerular Lesions | 69 |
| 4.7. EVALUATION OF CARCINOGENICITY..... | 70 |
| 4.7.1. Summary of Overall Weight of Evidence..... | 70 |
| 4.7.2. Synthesis of Human, Animal, and Other Supporting Evidence..... | 71 |
| 4.7.3. Mode-of-Action Information | 75 |
| 4.8. SUSCEPTIBLE POPULATIONS AND LIFE STAGES..... | 76 |
| 4.8.1. Possible Childhood Susceptibility | 76 |

| | |
|---|-----|
| 4.8.2. Possible Gender Differences..... | 76 |
| 5. DOSE-RESPONSE ASSESSMENTS..... | 78 |
| 5.1. ORAL REFERENCE DOSE (RfD)..... | 78 |
| 5.1.1. Choice of Principal Study and Critical Effect—with Rationale and Justification..... | 78 |
| 5.1.2. Methods of Analysis..... | 84 |
| 5.1.3. RfD Derivation—including Application of Uncertainty Factors (UFs)..... | 86 |
| 5.1.4. Reference value (RfV) Comparison Information..... | 87 |
| 5.1.5. Previous RfD Assessment..... | 90 |
| 5.2. INHALATION REFERENCE CONCENTRATION (RfC)..... | 90 |
| 5.3. CANCER ASSESSMENT..... | 91 |
| 5.3.1. Choice of Study/Data with Rationale and Justification..... | 91 |
| 5.3.2. Dose-Response Data..... | 92 |
| 5.3.3. Dose Adjustments and Extrapolation Methods..... | 94 |
| 5.3.4. Derivation of the Oral Cancer Slope Factor..... | 96 |
| 5.3.5. Uncertainties in Cancer Risk Values..... | 97 |
| 6. MAJOR CONCLUSIONS IN THE CHARACTERIZATION OF HAZARD AND DOSE RESPONSE..... | 102 |
| 6.1. HUMAN HAZARD POTENTIAL..... | 102 |
| 6.2. DOSE RESPONSE..... | 104 |
| 6.2.1. Noncancer..... | 104 |
| 6.2.2. Cancer..... | 106 |
| 7. REFERENCES..... | 108 |
| APPENDIX A. SUMMARY OF EXTERNAL PEER REVIEW AND PUBLIC COMMENTS AND DISPOSITION..... | A-1 |
| APPENDIX B. BENCHMARK DOSE CALCULATIONS FOR THE RfD..... | B-1 |
| APPENDIX C. TIME-TO-TUMOR MODELING RESULTS FROM TOX_RISK BASED ON THE INCIDENCE OF HEPATOCELLULAR CARCINOMAS..... | C-1 |

LIST OF TABLES

| | |
|--|----|
| Table 2-1. Physicochemical properties of chlordecone | 3 |
| Table 3-1. Whole blood chlordecone level by group of exposed subjects | 6 |
| Table 3-2. Distribution of chlordecone in exposed workers..... | 8 |
| Table 4-1. Incidence and time to tumor of hepatocellular carcinoma in rats | 24 |
| Table 4-2. Summary of endocrine and reproductive system tumor incidence among rats exposed to chlordecone..... | 25 |
| Table 4-3. Percent body weight gain and percent survival of chlordecone-exposed rats and mice..... | 28 |
| Table 4-4. Incidence and time to tumor of hepatocellular carcinoma in mice | 30 |
| Table 4-5. Testicular atrophy in male rats receiving chlordecone in the diet for 3 months | 32 |
| Table 4-6. Incidence of histopathologic liver lesions (fatty changes and hyperplasia) and renal glomerulosclerosis in male and female Wistar rats following administration of chlordecone in the diet for 1–2 years | 33 |
| Table 4-7. Effects of dietary chlordecone on reproduction in male and female mice (of mixed parentage) treated for 1 month prior to mating and for 100 days following the initiation of mating | 36 |
| Table 4-8. Effects of dietary chlordecone (0 or 40 ppm) on reproduction in BALB/cJaxGnMc mice during 100 days of treatment (preceded by 2 months of pre-mating treatment) and during 100 days of a crossover-mating period following the termination of treatment..... | 37 |
| Table 4-9. Effects of dietary chlordecone for 1 month prior to mating on reproductive indices of male and female laboratory mice of mixed breeds | 38 |
| Table 4-10. Effects of dietary chlordecone (0 or 5 ppm) 1 month prior to mating and up to 5 months after initiation of mating on reproduction in BALB/c mice..... | 39 |
| Table 4-11. Effects of chlordecone on estrous cyclicity and ovulation in CD-1 mice exposed to chlordecone by gavage 5 days/week for up to 6 weeks | 42 |
| Table 4-12. Abundance of various-sized follicles and the condition of large-sized follicles in the ovaries of female CD-1 mice exposed to chlordecone by gavage 5 days/week for 4 weeks | 43 |
| Table 4-13. Effects of chlordecone on adult female offspring of Sprague-Dawley rat dams administered chlordecone by gavage on GDs 14–20..... | 44 |

| | |
|--|-----|
| Table 4-14. Sperm parameters in male Sprague-Dawley rats following administration of chlordecone in the diet for 90 days..... | 45 |
| Table 4-15. Maternal and fetal effects following gavage dosing of pregnant rat dams with chlordecone on GDs 7–16..... | 47 |
| Table 4-16. Maternal and fetal effects following gavage dosing with chlordecone on GDs 7–16..... | 48 |
| Table 4-17. Physiochemical properties of chlordecone and mirex..... | 61 |
| Table 4-18. Summary of noncancer results for oral exposure studies of experimental animals to chlordecone..... | 64 |
| Table 5-1. Incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in male or female Wistar rats following administration of chlordecone in the diet for 1–2 years..... | 84 |
| Table 5-2. Possible PODs with applied uncertainty factors and resulting Potential RfVs..... | 88 |
| Table 5-3. Tumor incidence and time to first tumor for hepatocellular carcinomas observed in Osborne-Mendel rats following administration of chlordecone in the diet for 80 weeks..... | 93 |
| Table 5-4. Tumor incidence and time to first tumor for hepatocellular carcinomas observed in B6C3F ₁ mice following administration of chlordecone in the diet for 80 weeks..... | 93 |
| Table 5-5. Summary of time-to-tumor dose-response modeling based on the incidence of liver tumors in Osborne-Mendel rats and B6C3F ₁ mice..... | 97 |
| Table 5-6. Summary of uncertainty in the chlordecone cancer risk assessment..... | 98 |
| Table B-1. Incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in female Wistar rats following administration of chlordecone in the diet for 2 years..... | B-1 |
| Table B-2. BMD modeling results for the incidence of histopathologic renal lesions (glomerulosclerosis) in female Wistar rats, following administration of chlordecone in the diet for 2 years..... | B-2 |
| Table B-3. Incidence of testicular atrophy in male rats receiving chlordecone in the diet for 3 months..... | B-5 |
| Table B-4. BMD modeling results for the incidence of testicular atrophy in male Wistar rats, following administration of chlordecone in the diet for 3 months..... | B-6 |
| Table B-5. Incidence of histopathologic liver lesions (fatty changes and hyperplasia) in Wistar rats, following administration of chlordecone in the diet for 1–2 years..... | B-9 |

Table B-6. BMD modeling results for the increased incidence of liver lesions in rats (both sexes combined), following administration of chlordecone in the diet for 1–2 years..... B-10

Table B-7. Cauda Epididymal sperm count in male Sprague-Dawley rats receiving chlordecone in the diet for 3 months B-13

Table B-8. BMD modeling results for decreased epididymal sperm count in rats, following administration of chlordecone in the diet for 3 months B-13

LIST OF FIGURES

| | |
|--|------|
| Figure 2-1. The structure of chlordecone..... | 3 |
| Figure 3-1. A proposed metabolic scheme for chlordecone..... | 11 |
| Figure 4-1. Dosing regimen for male rats..... | 22 |
| Figure 4-2. Dosing regimen for female rats..... | 22 |
| Figure 4-3. Dosing regimen for male mice..... | 27 |
| Figure 4-4. Dosing regimen for female mice..... | 27 |
| Figure 5-1. Potential RfV comparison array for alternate points of departure..... | 89 |
| Figure B-1. Observed and predicted incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in female Wistar rats following administration of chlordecone in the diet for 1–2 years..... | B-2 |
| Figure B-2. Observed and predicted incidence of testicular atrophy in male Wistar rats, following administration of chlordecone in the diet for 3 months..... | B-6 |
| Figure B-3. Observed and predicted incidence of liver lesions in male and female Wistar rats following administration of chlordecone in the diet for 1–2 years..... | B-10 |
| Figure B-4. Observed and predicted epididymal sperm count in male rats, following administration of chlordecone in the diet for 3 months..... | B-14 |

LIST OF ABBREVIATIONS AND ACRONYMS

| | |
|--------------------|---|
| AIC | Akaike's Information Criterion |
| ALP | alkaline phosphatase |
| ALT | alanine aminotransferase |
| AST | aspartate aminotransferase |
| BMD | benchmark dose |
| BMD ₁₀ | benchmark dose associated with a 10% extra risk |
| BMDL ₁₀ | benchmark dose lower 95% confidence limit |
| BMDS | Benchmark Dose Software |
| BMR | benchmark response |
| BUN | blood urea nitrogen |
| CASRN | Chemical Abstracts Service Registry Number |
| CHO | Chinese hamster ovary |
| con A | concanavalin A |
| CYP450 | cytochrome P450 |
| DEN | diethylnitrosamine |
| EEG | electroencephalogram |
| ELISA | enzyme-linked immunosorbent assay |
| FSH | follicle-stimulating hormone |
| GGT | γ -glutamyl transpeptidase |
| GPT | glutamic pyruvic transferase |
| HDL | high-density lipoprotein |
| IRIS | Integrated Risk Information System |
| LD ₅₀ | median lethal dose |
| LOAEL | lowest-observed-adverse-effect level |
| LSPC | Life Science Products Company |
| MOA | mode of action |
| NCI | National Cancer Institute |
| NK | natural killer |
| NOAEL | no-observed-adverse-effect level |
| NRC | National Research Council |
| PBTK | physiologically based toxicokinetic |
| PFC | plaque-forming cell |
| PHA | phytohemagglutinin |
| PND | postnatal day |
| POD | point of departure |
| PVE | persistent vaginal estrus |
| RfC | reference concentration |
| RfD | reference dose |
| RfV | reference value |
| s.c. | subcutaneous |
| SER | smooth endoplasmic reticulum |
| SRBC | sheep red blood cell |
| STM | <i>Salmonella typhimurium</i> mitogen |
| TD | toxicodynamic |
| UF | uncertainty factor |
| U.S. EPA | U.S. Environmental Protection Agency |

FOREWORD

The purpose of this Toxicological Review is to provide scientific support and rationale for the hazard and dose-response assessment in IRIS pertaining to chronic exposure to chlordecone. It is not intended to be a comprehensive treatise on the chemical or toxicological nature of chlordecone.

The intent of Section 6, *Major Conclusions in the Characterization of Hazard and Dose Response*, is to present the major conclusions reached in the derivation of the reference dose, reference concentration, and cancer assessment, where applicable, and to characterize the overall confidence in the quantitative and qualitative aspects of hazard and dose response by addressing the quality of the data and related uncertainties. The discussion is intended to convey the limitations of the assessment and to aid and guide the risk assessor in the ensuing steps of the risk assessment process.

For other general information about this assessment or other questions relating to IRIS, the reader is referred to EPA's IRIS Hotline at (202) 566-1676 (phone), (202) 566-1749 (fax), or hotline.iris@epa.gov (email address).

AUTHORS, CONTRIBUTORS, AND REVIEWERS

CHEMICAL MANAGER/AUTHOR

Kathleen Newhouse, M.S.
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Washington, DC

CONTRIBUTING AUTHORS

Ted Berner, M.S.
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Washington, DC

Debdas Mukerjee, Ph.D.
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Cincinnati, OH

Andrew Rooney, Ph.D.
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Research Triangle Park, NC

CONTRACTING SUPPORT

Mark Follansbee, Ph.D.
Syracuse Research Corporation
N. Syracuse, NY

Julie Stickney, Ph.D.
Syracuse Research Corporation
N. Syracuse, NY

David Wohlers, Ph.D.
Syracuse Research Corporation
N. Syracuse, NY

REVIEWERS

This document has been reviewed by EPA scientists, interagency reviewers from other federal agencies, and the public, and peer reviewed by independent scientists external to EPA. A

summary and EPA's disposition of the comments received from the independent external peer reviewers and from the public is included in Appendix A.

INTERNAL EPA REVIEWERS

Chao Chen, Ph.D.
National Center for Environmental Assessment
Washington, DC

Harlal Choudhury, Ph.D.
National Center for Environmental Assessment
Cincinnati, OH

Lynn Flowers, Ph.D.
National Center for Environmental Assessment
Washington, DC

Karen Hogan M.S.
National Center for Environmental Assessment
Washington, DC

Prasada Kodavanti, Ph.D.
National Health and Environmental Effects Research Laboratory
Research Triangle Park, NC

Jamie Strong, Ph.D.
National Center for Environmental Assessment
Washington, DC

EXTERNAL PEER REVIEWERS

Harvey J. Clewell, Ph.D.
The Hamner Institutes for Health Sciences
Research Triangle Park, NC

George P. Daston, Ph.D.
Miami Valley Laboratories
The Proctor and Gamble Company
Cincinnati, OH

Gary L. Ginsberg, Ph.D. (Chair)
Connecticut Department of Public Health
Division of Environmental Epidemiology & Occupational Health
Hartford, CT

Michael I. Luster, Ph.D.
M. Luster Associates, LLC
Morgantown, WV

Lauren Zeise, Ph.D.
California Office of Environmental Health Hazard Assessment (OEHHA)
Oakland, CA

1. INTRODUCTION

This document presents background information and justification for the Integrated Risk Information System (IRIS) Summary of the hazard and dose-response assessment of chlordecone. IRIS Summaries may include oral reference dose (RfD) and inhalation reference concentration (RfC) values for chronic and other exposure durations, and a carcinogenicity assessment.

The RfD and RfC, if derived, provide quantitative information for use in risk assessments for health effects known or assumed to be produced through a nonlinear (presumed threshold) mode of action (MOA). The RfD (expressed in units of mg/kg-day) is defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. The inhalation RfC (expressed in units of mg/m³) is analogous to the oral RfD, but provides a continuous inhalation exposure estimate. The inhalation RfC considers toxic effects for both the respiratory system (portal of entry) and for effects peripheral to the respiratory system (extrapulmonary or systemic effects). Reference values are generally derived for chronic exposures (up to a lifetime), but may also be derived for acute (≤ 24 hours), short-term (>24 hours up to 30 days), and subchronic (>30 days up to 10% of lifetime) exposure durations, all of which are derived based on an assumption of continuous exposure throughout the duration specified. Unless specified otherwise, the RfD and RfC are derived for chronic exposure duration.

The carcinogenicity assessment provides information on the carcinogenic hazard potential of the substance in question and quantitative estimates of risk from oral and inhalation exposure may be derived. The information includes a weight-of-evidence judgment of the likelihood that the agent is a human carcinogen and the conditions under which the carcinogenic effects may be expressed. Quantitative risk estimates may be derived from the application of a low-dose extrapolation procedure. If derived, the oral slope factor is a plausible upper bound on the estimate of risk per mg/kg-day of oral exposure. Similarly, a plausible inhalation unit risk is an upper bound on the estimate of risk per $\mu\text{g}/\text{m}^3$ air breathed.

Development of these hazard identification and dose-response assessments for chlordecone has followed the general guidelines for risk assessment as set forth by the National Research Council (NRC, 1983). U.S. Environmental Protection Agency (U.S. EPA) Guidelines and Risk Assessment Forum Technical Panel Reports that may have been used in the development of this assessment include the following: *Guidelines for the Health Risk Assessment of Chemical Mixtures* (U.S. EPA, 1986a), *Guidelines for Mutagenicity Risk Assessment* (U.S. EPA, 1986b), *Recommendations for and Documentation of Biological Values for Use in Risk Assessment* (U.S. EPA, 1988), *Guidelines for Developmental Toxicity Risk*

Assessment (U.S. EPA, 1991), *Interim Policy for Particle Size and Limit Concentration Issues in Inhalation Toxicity* (U.S. EPA, 1994a), *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994b), *Use of the Benchmark Dose Approach in Health Risk Assessment* (U.S. EPA, 1995), *Guidelines for Reproductive Toxicity Risk Assessment* (U.S. EPA, 1996), *Guidelines for Neurotoxicity Risk Assessment* (U.S. EPA, 1998), *Science Policy Council Handbook: Risk Characterization* (U.S. EPA, 2000a), *Benchmark Dose Technical Guidance Document* (U.S. EPA, 2000b), *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (U.S. EPA, 2000c), *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002), *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA, 2005b), *Science Policy Council Handbook: Peer Review* (U.S. EPA, 2006a), and *A Framework for Assessing Health Risks of Environmental Exposures to Children* (U.S. EPA, 2006b).

The literature search strategy employed for this compound was based on the Chemical Abstracts Service Registry Number (CASRN) and at least one common name. Any pertinent scientific information submitted by the public to the IRIS Submission Desk was also considered in the development of this document. The relevant literature was reviewed through August 2009.

2. CHEMICAL AND PHYSICAL INFORMATION RELEVANT TO ASSESSMENTS

Chlordecone is a tan to white crystalline odorless solid (NIOSH, 2004). The structure of chlordecone is shown in Figure 2-1. Synonyms include Kepone, decachlorooctahydro-1,3,4-metheno-2H-cyclobuta[cd]-pentalen-2-one, and GC-1189 (O'Neil, 2001). Selected chemical and physical properties of chlordecone are listed in Table 2-1.

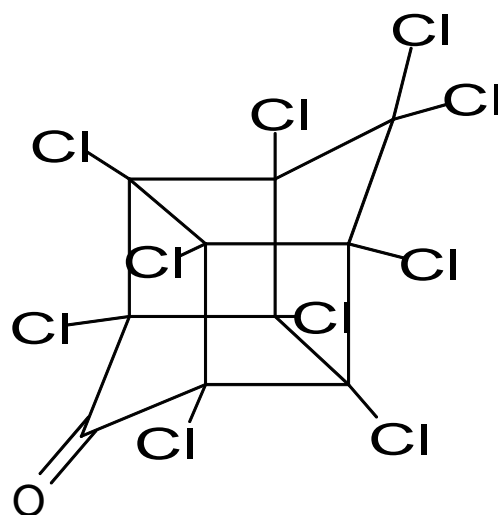


Figure 2-1. The structure of chlordecone.

Table 2-1. Physicochemical properties of chlordecone

| Characteristic | | Reference |
|-----------------------|---|---------------------|
| CAS number | 143-50-0 | Lide, 2000 |
| Molecular weight | 490.64 | O'Neil, 2001 |
| Chemical formula | C ₁₀ Cl ₁₀ O | O'Neil, 2001 |
| Melting point | Decomposes at 350°C | Lide, 2000 |
| Vapor pressure | 2.25 × 10 ⁻⁷ mm Hg at 25°C | Kilzer et al., 1979 |
| Density | 1.61 g/mL at 25°C | Lide, 2000 |
| Water solubility | 2.70 mg/L at 25°C | Kilzer et al., 1979 |
| Other solubilities | Slightly soluble in hydrocarbon solvents; soluble in alcohols, ketones, acetic acid | O'Neil, 2001 |
| Partition coefficient | log K _{ow} = 5.41 | Hansch et al., 1995 |

Chlordecone production begins with the condensation of hexachlorocyclopentadiene with sulfur trioxide under heat and pressure (NLM, 2004a; ATSDR, 1995). Antimony pentachloride is used as a catalyst. The product of this reaction is hydrolyzed and then neutralized (ATSDR, 1995; IARC, 1979). Chlordecone is obtained by centrifugation or filtration and hot air drying.

Chlordecone is also a contaminant in mirex formulations and is a degradation product of mirex (Bus and Leber, 2001).

Chlordecone was first produced in the United States in the early 1950s (IARC, 1979). It was introduced commercially in 1958 (Bus and Leber, 2001). Approximately 3.6 million pounds of chlordecone were produced in the United States between 1951 and 1975 (ATSDR, 1995). Chlordecone production in the United States ended in 1975 after intoxication from severe industrial exposure was observed in employees who worked at the only chlordecone manufacturing plant in the country (Bus and Leber, 2001). Typical signs of chlordecone intoxication include nervousness, headache, and tremor (Cannon et al., 1978).

Chlordecone was primarily used as an insecticide (IARC, 1979). Specific applications have included control of the banana root borer, application on non-fruit-bearing citrus trees to control rust mites, control of wireworms in tobacco fields, control of apple scab and powdery mildew, control of the grass mole cricket, and control of slugs, snails, and fire ants (NLM, 2004a; ATSDR, 1995). Its registration was cancelled in 1978 (Metcalf, 2002; IARC, 1979).

Chlordecone is resistant to degradation in the environment. It is not expected to react with hydroxyl radicals in the atmosphere or to hydrolyze or photolyze (NLM, 2004a). Chlordecone in the air is likely to be removed by deposition of particles (NLM, 2004a). Studies have shown that microorganisms degrade chlordecone slowly (NLM, 2004a). Chlordecone is expected to adsorb to soil and to stick to suspended solids and sediments in water (NLM, 2004a). Small amounts of chlordecone will evaporate from soil or water surfaces (NLM, 2004a). Chlordecone has a high potential for bioaccumulation in fish and other aquatic organisms (ATSDR, 1995).

3. TOXICOKINETICS

The available data for humans and animals indicate that chlordecone is well absorbed following oral exposure. Once absorbed, it is widely distributed and eventually concentrates in the liver. It is metabolized by humans and some animal species to chlordecone alcohol. Glucuronide conjugates of chlordecone and chlordecone alcohol, as well as unconjugated chlordecone, are slowly excreted in the bile and eliminated in the feces. Fecal excretion is limited by enterohepatic recirculation.

3.1. ABSORPTION

Chlordecone absorption in humans has been demonstrated by the measurement of chlordecone concentrations in blood, subcutaneous (s.c.) fat, and other body fluids and tissues following subchronic occupational exposure, presumably through ingestion, inhalation, and dermal contact (Taylor, 1982; Adir et al., 1978; Cannon et al., 1978; Cohn et al., 1978). Workers categorized as having subjective or objective neurological symptoms of chlordecone toxicity (i.e., nervousness, tremulousness, ataxia) had whole blood concentrations ranging between 0.009 and 11.8 ppm (Cannon et al., 1978). Workers with subjective symptoms alone represented 36% of identified cases. Chlordecone blood levels of the subset of workers with clinically confirmed neurological symptoms were not reported. Chlordecone blood concentrations for workers without neurological symptoms were between 0.003 and 4.1 ppm. Chlordecone was also detected in the blood of Hopewell community residents living near a pesticide plant with concentrations ranging from 0.005 to 0.0325 ppm. Potential exposure routes for community residents included inhalation of chlordecone associated with fine particulate matter and ingestion of contaminated soil and drinking water. Neurological symptoms were reported by some residents living near the plant site. In general, the highest blood chlordecone concentrations were observed in affected workers, and lower concentrations were measured in unaffected workers and community residents (Table 3-1) (Cannon et al., 1978).

Table 3-1. Whole blood chlordecone level by group of exposed subjects

| Group | Number tested | Number with detectable level | Percent with detectable level | Range of detectable level, ppm | Mean of detectable level, ppm |
|---|---------------|------------------------------|-------------------------------|--------------------------------|-------------------------------|
| Affected LSPC ^a workers | 57 | 57 | 100 | 0.009–11.8 | 2.53 |
| Unaffected LSPC workers | 49 | 48 | 99 | 0.003–4.1 | 0.60 |
| Family members, LSPC workers | 32 | 30 | 94 | 0.003–0.39 | 0.10 |
| Allied ^b chlordecone workers | 39 | 30 | 77 | 0.003–0.45 | 0.06 |
| Neighborhood workers | 32 | 23 | 72 | 0.003–0.031 | 0.011 |
| Sewage treatment plant workers | 10 | 6 | 60 | 0.004–0.014 | 0.006 |
| Cab drivers | 5 | 1 | 20 | 0.003 | 0.003 |
| Truck drivers | 2 | 1 | 50 | 0.004 | 0.004 |
| Hopewell community residents ^c | 214 | 40 | 19 | 0.005–0.0325 | 0.011 |

^aLSPC = Life Science Products Company workers with self reported or clinically observed neurological symptoms.

^bAllied Chemical Corporation.

^cExcludes chlordecone factory workers.

Source: Cannon et al. (1978).

No data were available in laboratory animals to evaluate chlordecone absorption following inhalation exposure. Quantitative data on absorption of orally administered chlordecone are limited; however, studies on the distribution and excretion of chlordecone in rats, mice, gerbils, and pigs following oral administration of chlordecone indicate that this chemical is readily absorbed from the gastrointestinal tract in animals (Hewitt et al., 1985; Aldous et al., 1983; Fujimori et al., 1982a; Wang et al., 1981; Kavlock et al., 1980; Egle et al., 1978). One study (Egle et al., 1978) attempted to estimate oral absorption quantitatively. Male Sprague-Dawley rats received a single oral dose of 40 mg/kg-day [¹⁴C]-labeled chlordecone in corn oil solution. The percentage of radioactivity excreted in the feces was measured over time. Approximately 10% of the dose was detected in the feces on the first day after dosing, suggesting that 90% of the orally administered dose was absorbed from the corn oil vehicle.

Animal studies suggest that chlordecone is absorbed only to a limited extent through the skin (Heatherington et al., 1998; Shah et al., 1987). The in vivo percutaneous absorption of chlordecone was evaluated in young (33 days old) and adult (82 days old) F344 rats (Shah et al., 1987). Acetone solution that contained [¹⁴C]-labeled chlordecone was applied to the shaved backs of animals, with the treatment area constituting 2–3% of the total body surface area.

Urine and feces were collected over a 72-hour period, after which animals were sacrificed to determine the recovery of radioactivity and the percutaneous absorption of chlordecone.

Three dose levels were used to compare dermal absorption in young and adult rats (three rats/dose group). No age-related differences in dermal absorption of chlordecone were noted in this study. Dermal absorption decreased over the dose range in both young and adult rats. In adults, 9% of the applied dose was absorbed at the lowest dose ($0.26 \mu\text{mol}/\text{cm}^2$), 6% absorption occurred at the medium dose ($0.54 \mu\text{mol}/\text{cm}^2$), and 1% absorption occurred at the highest dose tested ($2.68 \mu\text{mol}/\text{cm}^2$). In young rats, 10% of the applied dose was absorbed at the lowest dose ($0.34 \mu\text{mol}/\text{cm}^2$), 7% absorption occurred at the medium dose ($0.54 \mu\text{mol}/\text{cm}^2$), and 2% absorption was seen at the highest dose tested ($2.68 \mu\text{mol}/\text{cm}^2$). The nonlinear relationship between in vivo dermal absorption and dose described by Shah et al. (1987) was confirmed by Heatherington et al. (1998) in young and adult rats. In adults, 8% of the applied dose was absorbed at the lowest dose ($0.29 \mu\text{mol}/\text{cm}^2$), 6% absorption was seen at the medium dose ($0.54 \mu\text{mol}/\text{cm}^2$), and 1% absorption occurred at the highest dose tested ($2.68 \mu\text{mol}/\text{cm}^2$). In young rats, 9% of the applied dose was absorbed at the lowest dose ($0.34 \mu\text{mol}/\text{cm}^2$), 7% absorption occurred at the medium dose ($0.54 \mu\text{mol}/\text{cm}^2$), and 2% absorption was seen at the highest dose tested ($2.68 \mu\text{mol}/\text{cm}^2$).

The time course of chlordecone dermal absorption was studied in young and adult rats by using a serial sacrifice study design (Heatherington et al., 1998). Young and adult F344 rats were dermally exposed to $0.285 \mu\text{mol}/\text{cm}^2$ chlordecone by using the procedure described above for the Shah et al. (1987) study. Rats were sacrificed at 6, 24, 48, 72, and 120 hours posttreatment. No significant age-related differences were noted in the time course for dermal penetration of chlordecone. In adult rats, the average cumulative absorption was 0.4, 3, 6, 9, and 14% measured at 6, 24, 48, 72, and 120 hours, respectively. In young rats, the average cumulative absorption was 0.6, 4, 7, 10, and 14% measured at 6, 24, 48, 72, and 120 hours, respectively. In vitro test systems using static and flow through diffusion cells were also employed by Heatherington et al. (1998). Only 1% of the applied chlordecone dose penetrated excised dorsal skin from young and adult rats under in vitro conditions. Based on the in vivo dermal absorption data obtained, a biophysically based percutaneous absorption model was developed to describe the movement of chlordecone through the skin. This model was embedded in a whole animal physiologically based toxicokinetic (PBTK) model that was employed to predict tissue concentrations of chlordecone following dermal exposure (see Section 3.5).

3.2. DISTRIBUTION

In 32 workers exposed to chlordecone for a period that ranged from 3 to 16 months, high concentrations of chlordecone were found in blood, liver, and s.c. fat. Modest amounts of chlordecone were detected in muscle, gall bladder, bile, and stool, while only trace amounts were detected in aqueous body fluids such as cerebrospinal fluid, urine, saliva, and gastric juice (Cohn et al., 1978). The ratio of the chlordecone concentration in fat as compared to the chlordecone

concentration in the blood was 7:1, which is relatively low for a lipophilic organochlorine pesticide. The liver to blood concentration ratio in exposed workers was reported to be 15:1 (Table 3-2).

Table 3-2. Distribution of chlordecone in exposed workers

| Tissue | Number of patients | Concentration range (µg/g) | Partition | |
|------------------|--------------------|----------------------------|--------------|---------|
| | | | Tissue:blood | Range |
| Whole blood | 32 | 0.6–32.0 | 1.0 | |
| Liver | 10 | 13.3–173.0 | 15.0 | 4.6–31 |
| Subcutaneous fat | 29 | 1.7–62.1 | 6.7 | 3.8–12 |
| Muscle | 5 | 1.2–11.3 | 2.9 | 1.8–4.5 |
| Gallbladder bile | 6 | 2.5–30.0 | 2.5 | 1.4–4.1 |

Source: Cohn et al. (1978).

The preferential uptake and slow elimination of chlordecone from the liver was confirmed in laboratory animals (Belfiore et al., 2007; Hewitt et al., 1985; Egle et al., 1978). Chlordecone concentrations in rat plasma, kidney, liver, and adipose tissue were determined at various time points following a single oral dose of 50 mg/kg (Hewitt et al., 1985). Chlordecone concentrations persisted in rat tissues throughout the 32-day study period. The highest tissue concentrations were observed in the liver, and this organ had the slowest elimination rate. Between days 8 and 32, liver concentrations were reduced by 73%, while plasma, kidney, and adipose levels were reduced 90, 88, and 81%, respectively. The distribution of chlordecone was also studied in rats receiving a single oral dose of 40 mg/kg-day [¹⁴C]-labeled chlordecone in corn oil solution (Egle et al., 1978). Initially, the highest levels of radioactivity were found in the adrenal glands followed by liver, lung, and fat. By 3 days following dosing, the highest concentration was in the liver, and this continued throughout the 182-day study period. Chlordecone is eliminated more slowly from the liver as compared with other tissues. The liver to blood ratio increased from 28:1 on day 1 to 126:1 on day 84. The fat to blood ratio reached a maximum of 31:1 on day 7 and declined thereafter, while other organ to blood ratios remained constant. Belfiore et al. (2007) measured chlordecone concentrations in rat liver, fat, blood, kidney, and muscle at 1, 14, or 30 days following a single oral dose of 40 mg/kg. The highest tissue concentrations were observed in the liver, followed by the kidney. The slowest elimination rate was seen in the liver, with chlordecone concentrations reduced 25% between day 1 and day 30. At day 30, levels were reduced by 65, 69, 73, and 75% in blood, fat, muscle, and kidney, respectively. Liver to blood ratios increased from 71:1 on day 1 to 150:1 on day 30.

The preferential retention of chlordecone by the liver is related to chlordecone binding to plasma proteins and lipoproteins. Serum gel filtration indicated that chlordecone was predominantly bound to albumin and lipoproteins in exposed workers. Electrophoresis of

normal human plasma following the addition of [¹⁴C]-labeled chlordecone demonstrated 80% binding to lipoproteins, with most of this binding associated with high-density lipoproteins (HDLs) (Skalsky et al., 1979). The preferential binding of chlordecone to albumin and HDL was demonstrated in human, rat, and pig plasma (Soine et al., 1982). In human plasma, the in vitro distribution of [¹⁴C]-labeled chlordecone was 46% protein, 30% HDL, 20% low density lipoprotein, and 6% very low density lipoprotein. Similar distributions were seen for pig plasma and for in vitro and in vivo distribution studies in rat plasma. Albumin was identified as the major component of the protein fraction that binds chlordecone. Experiments in isolated perfused pig liver demonstrated that an increase in HDL can affect the distribution of chlordecone, favoring chlordecone uptake and retention in the liver and decreased chlordecone elimination in the bile (Soine et al., 1984). Chlordecone and cholesterol have been shown to compete for similar intracellular binding and transport proteins, which are inducible by chlordecone pretreatment (Gilroy et al., 1994; Carpenter and Curtis, 1991, 1989).

The brain and plasma levels of chlordecone in mice were measured after daily oral dosing with 10 or 50 mg/kg-day (Wang et al., 1981). At the lower dose, the plasma level of chlordecone increased steadily throughout the 12-day treatment period, while the brain chlordecone level reached a plateau on day 10. Brain and plasma levels decayed biphasically following administration of 50 mg/kg-day chlordecone for 1 or 2 days. Brain and plasma concentrations were correlated with loss of motor control at both administered dose levels. Chlordecone was distributed to discrete areas of the mouse brain following a single gavage dose of 50 mg/kg (Fujimori et al., 1982a). The striatum and the medulla/pons had significantly higher chlordecone levels than the cortex, midbrain, or cerebellum.

The distribution of chlordecone following dermal absorption was studied by Heatherington et al. (1998) in young and adult rats (see Section 3.1 for study design information). Less than 15% of the applied dose was absorbed within 120 hours. Organ concentrations increased slowly over time, with the highest concentrations observed in the liver followed by (in decreasing order) kidney, carcass, skin, and blood. Kinetic differences in liver accumulation of chlordecone were suggested between young and adult rats, but all other organ concentrations were comparable. Tissue levels did not appear to have reached steady-state conditions by 120 hours of dermal exposure to chlordecone.

Kavlock et al. (1980) studied the distribution of chlordecone in fetal and neonatal rats. Pregnant rats were given an oral dose of 5 mg/kg chlordecone on gestation days (GDs) 15, 18, or 20. For the prenatal study, animals were killed at 4, 24, or 48 hours after dosing, and maternal and fetal tissues were obtained for chlordecone analysis. In the postnatal study, the dams were given chlordecone at a dose of either 1 or 10 mg/kg-day on days 2–5 of the lactation period. Maternal milk was obtained following an injection of oxytocin on GDs 5, 9, and 15. Pups were sacrificed for chlordecone tissue analysis on days 3, 5, 7, 9, 12, 15, and 17 of lactation. Chlordecone crossed the placenta and was observed in fetal tissues as early as 4 hours after

maternal dosing. The maximum concentrations of chlordecone on the placenta were 3.5 and 4.0 ppm. Maternal tissue levels were 4 to 5 times higher than fetal concentrations, indicating some retardation in distribution of chlordecone to the fetus. Chlordecone levels in the fetus were highest in the liver, followed by the brain, heart, and kidneys. Chlordecone excretion into milk was an important pathway for elimination in nursing dams. Neonatal organ concentrations of chlordecone increased steadily over the lactation period. Tissue uptake for neonates was highest in the liver, followed by the brain and the eyes. Day 5 liver and brain levels rose from 2 to 23 μg and from 16 to 150 μg , respectively, in pups nursed by 10 mg/kg-day dosed dams. Tissue concentrations were correlated with chlordecone levels in milk.

The tissue distribution of chlordecone was investigated in rats following pretreatment with phenobarbital, an inducer of hepatic metabolism (Aldous et al., 1983). Repeat doses of phenobarbital (65 mg/kg) were administered intraperitoneally to adult male Sprague-Dawley rats 6, 12, and 24 hours prior to gavage administration of [^{14}C]-labeled chlordecone. Phenobarbital pretreatment resulted in an increase in the specific activity in the liver and uniformly reduced the specific activity in other tissues. In phenobarbital pretreated rats, 87% of the [^{14}C]-labeled chlordecone was found in the liver, compared to 55% in control rats not receiving phenobarbital. Fecal and urinary excretion of chlordecone was reduced. A single dose of phenobarbital (12 or 24 hours prior to chlordecone administration) similarly altered the distribution of chlordecone; however, changes were more marked with multiple dose administration.

3.3. METABOLISM

Based on available data, a proposed metabolic scheme for chlordecone is shown in Figure 3-1. Although chlordecone is not extensively metabolized in mammals, chlordecone alcohol is formed in humans and some laboratory animal species by reduction of the hydrated carbonyl group (Fariss et al., 1980; Blanke et al., 1978). A cytosolic aldo-keto reductase enzyme appears to be responsible for the formation of chlordecone alcohol (Molowa et al., 1986). Chlordecone alcohol is excreted in bile primarily as a glucuronide conjugate, while chlordecone is excreted into bile mostly in the unconjugated form (Fariss et al., 1980).

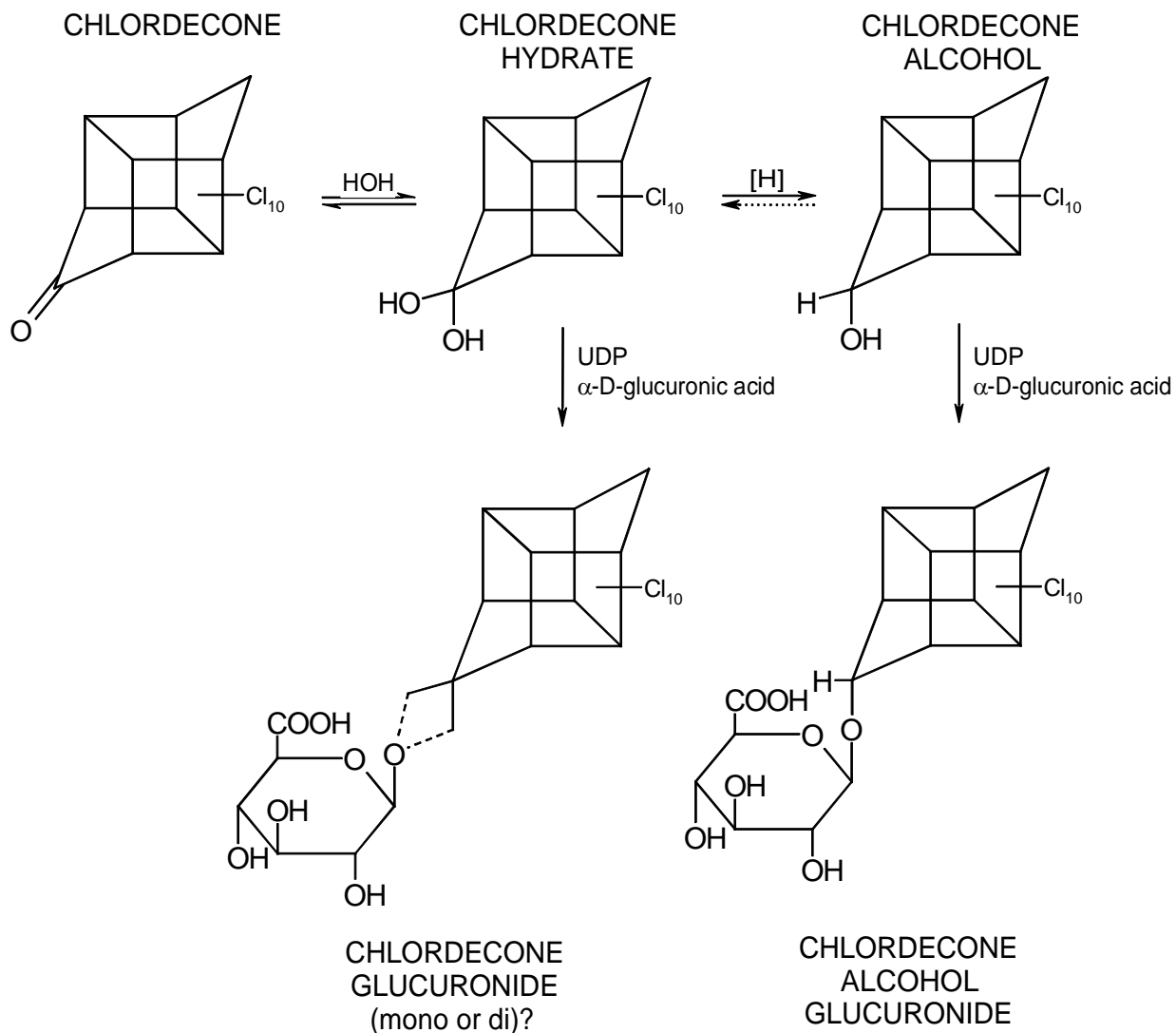


Figure 3-1. A proposed metabolic scheme for chlordecone.

The metabolism of chlordecone to chlordecone alcohol occurs in humans, gerbils, and pigs but not to a significant extent in rats, mice, guinea pigs, or hamsters (Houston et al., 1981; Fariss et al., 1980; Blanke et al., 1978). Species differences were also observed in phase II conjugation reactions, with chlordecone conjugation occurring in humans but not in gerbils or rats (Houston et al., 1981). In humans, a reduced form of chlordecone was first identified in the stool of pesticide workers experiencing symptoms of chlordecone toxicity, including nervousness, headache, and tremor (Blanke et al., 1978). Fariss et al. (1980) utilized human bile samples for further analysis of chlordecone and possible metabolites. Human bile was obtained from exposed workers by either aspirated duodenal contents (six workers) or directly from a T-tube that was implanted during gallbladder surgery (one worker). The initial analysis of human bile using gas-liquid chromatography revealed significant amounts of free chlordecone

and small amounts of free chlordecone alcohol in exposed workers. Subsequent treatment of bile samples with β -glucuronidase prior to the analysis resulted in large amounts of measurable chlordecone alcohol. It was estimated that >90% of the chlordecone alcohol in human bile is present as a glucuronide conjugate, while <10% of the chlordecone parent compound is conjugated prior to biliary excretion. The ratio of chlordecone to chlordecone alcohol following β -glucuronidase, sulfatase, and acid hydrolysis treatments was between 1:2 and 1:4 in human bile. In contrast, rat bile contained only trace amounts of chlordecone alcohol, with a corresponding chlordecone to chlordecone alcohol ratio of 155:1.

Molowa et al. (1986) characterized a unique cytosolic aldo-keto reductase enzyme responsible for the conversion of chlordecone to chlordecone alcohol. Chlordecone reductase activity was detected in the liver cytosol of rabbits, gerbils, and humans but was absent in rats, mice, hamsters, and guinea pigs. Pretreatment of gerbils with a single oral dose of chlordecone (20 mg/kg) resulted in a 38% increase in the specific activity of chlordecone reductase 7 days later. Soine et al. (1983) also demonstrated the metabolism of chlordecone to chlordecone alcohol in the pig. Pigs were given an intraperitoneal dose of either 40 or 80 mg/kg-day, and chlordecone and chlordecone alcohol concentrations in the blood and gallbladder bile were measured at regular intervals over a 35-day study period. At the end of the study, hepatic bile, liver, and feces were also analyzed for chlordecone and chlordecone alcohol levels. The plasma half-life of chlordecone in the pig was determined to be 12 days at the higher dose and 22 days at the lower dose. Chlordecone metabolites were generally not detected in the plasma; however, free chlordecone, free chlordecone alcohol, and conjugated chlordecone alcohol were measured in gallbladder bile at both doses. Conjugated chlordecone was only observed in gallbladder bile at the high-dose level. The induction of chlordecone reductase in the pig was suggested by the observed increase in the chlordecone alcohol to chlordecone ratio in the gallbladder bile over the time course of the study. On the last day of the study, 20% of chlordecone was conjugated in the plasma and bile, while only 3% of chlordecone was conjugated in the liver and feces. Chlordecone alcohol was not detected in the plasma or the liver, but was 85% conjugated in the bile and 15% conjugated in the feces.

Chlordecone has been shown to induce the cytochrome 450 (CYP450) mixed function oxidase enzyme system in male and female rats (Gilroy et al., 1994; Hewitt et al., 1985; Mehendale et al., 1978, 1977). Mehendale et al. (1978, 1977) exposed male and female rats to 0, 50, 100, or 150 ppm chlordecone in the diet for 16 days. A dose-related decrease in body weight gain was observed, while liver weights were unaltered by chlordecone treatment. Enzyme activities that were increased by chlordecone treatment at each dose level included aniline, pentobarbital, and hexobarbital hydroxylation, and aminopyrine and ethylmorphine demethylation. CYP450, cytochrome c reductase, and aniline binding were all increased, while cytochrome b5 and NADPH dehydrogenase activity were unaffected by chlordecone treatment. Hewitt et al. (1985) demonstrated increases in microsomal CYP450 and NADPH cytochrome c

reductase following a single oral dose of 50 mg/kg (days 2 to 32). Cytochrome b5 was also increased, but not until 24 to 32 days after chlordecone administration. A single oral dose of 15 mg/kg to Sprague-Dawley rats resulted in an increase in CYP450 and ethoxyresorufin-O-deethylase and ethoxycoumarin-O-deethylase enzyme activities (Gilroy et al., 1994). Weanling pups of Sprague-Dawley rat dams exposed to chlordecone from GD 2 to day 21 postpartum (0, 0.1, 1, or 1.5 mg/kg-day) exhibited a dose-related increase in metabolism and excretion of lindane (Chadwick et al., 1979).

Chlordecone was shown to selectively induce CYP2B2 in adult rat hepatocyte cultures (Kocarek et al., 1991). Chlordecone selectively increased the mRNA for CYP2B2, and both chlordecone and chlordecone alcohol induced the immunoreactive protein levels for CYP2B2. Chlordecone did not affect the mRNA or immunoreactive protein levels for CYP2B1 in isolated rat hepatocytes. In addition to its selective induction of CYP2B2, chlordecone also suppressed the induction of CYP2B1 and CYP2B2 when coincubated with phenobarbital in hepatocyte culture. Mechanistic studies suggest that selective induction of CYP2B2 is not due to the estrogenic properties of chlordecone, while the ability to suppress phenobarbital induction may relate to the gem-diol configuration of chlordecone (Kocarek et al., 1994).

3.4. ELIMINATION

Chlordecone and chlordecone alcohol are eliminated from the body primarily through biliary excretion into feces. In humans, chlordecone is eliminated slowly from the blood. Estimates of the chlordecone serum half-life ($t_{1/2}$) in chemical plant workers ranged from 63 to 128 days (Adir et al., 1978). Analysis of excretory fluids in exposed pesticide workers showed that, while chlordecone was undetectable in sweat and present only in minor quantities in urine, saliva, and gastric juice concentrations in gallbladder bile were approximately equivalent to chlordecone concentrations in blood (Cohn et al., 1978). The excretion rate of chlordecone into hepatic bile was estimated from either aspirated duodenal contents (six workers) or bile collected directly from a T-tube that was implanted during gallbladder surgery (one worker) (Cohn et al., 1978). The biliary excretion rates varied widely among workers (~1–10 mg/day); however, the daily excretion amount expressed as a percent of the total body content was relatively constant (0.29–0.85%). For workers who underwent duodenal aspiration, only 5–10% of the chlordecone that entered the duodenal lumen via the bile was detected in the feces. Similarly, the rate of chlordecone excreted in bile collected from a surgically implanted T-tube was 19 times greater than the rate of elimination in the stool. These results suggest that enterohepatic recycling plays an important role in the slow excretion of chlordecone. In order to prevent the reabsorption of chlordecone into the gastrointestinal tract, cholestyramine was investigated as a possible treatment for chlordecone intoxication. Cholestyramine is an anion-exchange resin that binds chlordecone but is not absorbed in the gastrointestinal tract. Treatment with cholestyramine reduced the average $t_{1/2}$ in the blood of workers from 165 to 80 days (Cohn et al., 1978).

Gastrointestinal secretion of chlordecone also appears to play a role in fecal excretion in humans (Boylan et al., 1979). Diversion of the bile stream from the intestine was accomplished in a chlordecone-exposed worker with a surgically implanted T-tube. Chlordecone excretion in stool increased eightfold when bile was diverted from the gut. This nonbiliary mechanism for fecal excretion does not appear to be related to salivary or gastric juice, because chlordecone concentrations in these fluids were minimal in exposed workers. Chlordecone is transferred from the bloodstream to gastrointestinal lumen via a secretory process governed by diffusion (Bungay et al., 1979). High concentrations of chlordecone in the lumen inhibit gastrointestinal secretion. Experimental data in rats confirmed the presence of a nonbiliary pathway for fecal excretion of chlordecone. Bungay et al. (1979) evaluated the transport of chlordecone in and out of the gut and utilized a PBTK model to describe the results (see Section 3.5). The transport of chlordecone into and out of the gut was studied following intravenous administration to the bile duct of cannulated rats and oral administration to intact rats.

Animal studies evaluated the elimination of chlordecone following oral exposure. Egle et al. (1978) studied chlordecone excretion in male Sprague-Dawley rats receiving a single oral dose of 40 mg/kg-day [¹⁴C]-labeled chlordecone in corn oil solution. The percentage of radioactivity excreted in the feces was measured over time. Approximately 30% of the administered chlordecone was excreted within the first 7 days, after which the rate of excretion steadily declined. After 12 weeks, 65.5% of the dose had been excreted into the feces and after 26 weeks, the cumulative excretion in feces was only 69.8%. A small amount of the administered chlordecone was excreted in the urine. Only 1.6% of the administered dose was found in the urine by 12 weeks, one-third of which was excreted into urine in the first 24 hours. Chlordecone was measured in expired air on days 1 and 9 after dosing, and less than 1% of the administered dose was detected in expired air.

Heatherington et al. (1998) studied the excretion of chlordecone following dermal absorption in young and adult rats (see Section 3.1 for study methods). Higher concentrations of chlordecone were detected in the urine of young rats as compared with adults. Chlordecone elimination was primarily in the feces, with limited urinary excretion. Feces to urine ratios 120 hours following dermal application of chlordecone were 3:1 and 3:8 in young and adult rats, respectively.

Chlordecone treatment has been shown to decrease the biliary excretion of other chemicals (Curtis and Mehendale, 1979). Male Sprague-Dawley rats were fed diets containing 0, 10, 50, or 150 ppm chlordecone for 15 days. Food consumption and body weight data were used to estimate daily dose levels of 0, 0.69, 3.2, and 8.0 mg/kg-day. Clinical signs of chlordecone toxicity were not apparent in the 10 or 50 ppm groups, but hyperexcitability and tremors were observed at 150 ppm. Decreased body weight gain was observed at the two highest dose levels. Biliary function was evaluated in bile-duct-cannulated intact animal preparations. The highest dose of chlordecone reduced the biliary excretion of the polar metabolites of

imipramine (31% of control) and phenolphthalein glucuronide (27% of control). These decreases occurred despite an increase in cumulative bile flow at the 150 ppm dose level. Oligomycin-sensitive mitochondrial ATPase activity was inhibited by chlordecone in this study; however, the dose-response data do not suggest a direct correlation between enzyme inhibition and hepatobiliary dysfunction.

Teo and Vore (1991) studied the effect of chlordecone on bile acid secretory function (i.e., bile flow, bile acid concentration, bile acid secretory rate) in the isolated perfused rat liver. Rats were given an oral dose of 18.75 mg/kg-day chlordecone for 3 days prior to measurement of bile secretory parameters. Chlordecone treatment resulted in an increase in bile flow, but a decrease in bile acid concentration and bile acid secretory rate. These results suggest that chlordecone acts primarily at the bile canalicular membrane to decrease biliary excretion. Rochelle et al. (1990) demonstrated that chlordecone perturbs the membrane and inhibits the active transport of glutamate at the bile canalicular membrane. Hepatobiliary dysfunction does not appear to be related to the concentration of chlordecone associated with the liver plasma membrane (Rochelle and Curtis, 1994); however, inhibition and recovery of 5'-nucleotidase activity in the liver plasma membrane suggest that biochemical alterations in membrane function may be involved.

3.5. PHYSIOLOGICALLY BASED TOXICOKINETIC MODELS

PBTK models have been used to describe the hepatic sequestration of chlordecone (Belfiore et al., 2007), movement of chlordecone in and out of the gut (Bungay et al., 1979), percutaneous absorption and disposition of chlordecone (Heatherington et al., 1998), and toxic interactions between chlordecone and carbon tetrachloride (el-Masri et al., 1995) in laboratory animals. PBTK models are not available to describe toxicokinetic processes in humans.

Belfiore et al. (2007) developed a PBTK model to describe sequestration of chlordecone in the liver of rats. Male Sprague-Dawley rats received a one-time treatment of 40 mg/kg-day of chlordecone in corn oil by gavage. Rats were sacrificed at 1, 14, or 30 days following dosing, and liver, fat, kidney, and muscle specimens were removed and assayed for chlordecone concentration. Data from this time course and from distribution studies in the literature (Hewitt et al., 1985; Egle et al., 1978) were used to develop and validate a toxicokinetic model to describe the preferential sequestration of chlordecone in the liver. A model was constructed in which liver, fat, and slowly perfused and rapidly perfused tissues were flow limited. Metabolism was not included due to the low biotransformation rate for chlordecone. The model fit to the experimental data was greatly improved by adding blood and liver binding coefficients derived from data from Soine et al. (1984, 1982). This model provides additional support for the hepatic sequestration of chlordecone in Sprague-Dawley rats; however, several factors limit its use in the derivation of reference values. It is not known how the measured blood, fat, or liver tissue levels would correlate other organ compartments not included in the model. This model also does not

provide information on inhalation exposure that would be needed for route-to-route extrapolation. Additionally, the model is not parameterized for humans, so it cannot be used to evaluate interspecies toxicokinetic differences.

Bungay et al. (1979) conducted experiments comparing intravenous administration of chlordecone in bile-duct-cannulated rats and oral administration in intact rats. The data were used in the gut portion of a whole body PBTK model. The gastrointestinal tract was divided into six segments, and the lumens of these segments were connected in series in the model. Flow rates were measured in each segment, and the net secretion or absorption was determined for each compartment. Diffusional processes were assumed to govern chlordecone exchange between blood, gut tissue, and the lumen. In the rat, the PBTK model yields a maximum clearance estimate for gut secretion of 25 mL/hour. Measurement of biliary clearance in bile-duct-cannulated rats was 5 mL/hour, suggesting a total maximum clearance rate of 30 mL/hour. Assuming that the permeability of the gut to chlordecone is similar in rats and humans, the authors calculated a maximum human clearance rate of 1,000 mL/hour by using a body-weight scaling factor (body-weight ratio raised to the 2/3 power). The chlordecone clearance rate estimated for pesticide workers not receiving cholestyramine treatment (Cohn et al., 1978) was only 40 mL/hour due to the presence of chlordecone in the lumens and the inhibition of diffusion from the gut.

A PBTK model was developed to describe the percutaneous absorption and disposition of chlordecone in young and adult rats (Heatherington et al., 1998). The experimental data for the dose effect and time course of chlordecone dermal absorption are described in Section 3.1. The distribution and excretion data for this study are reported in Sections 3.2 and 3.3. A biophysically-based percutaneous absorption model was developed based on in vivo dermal absorption data. The absorption model consisted of five first-order rate constants describing the movement of chlordecone by diffusion from the site of application to the stratum corneum, where it undergoes partitioning with the viable epidermis, followed by entry into the blood and distribution throughout the body. The rate constants for movement among compartments were based on chlordecone physical and chemical characteristics, skin physiology, and experimental data. The absorption model was significantly limited by its inability to describe the nonlinear dose effect of percutaneous exposure (i.e., decreasing percent absorption with increasing dose). Therefore, the data for only one dose level could be used for PBTK disposition modeling (i.e., time course data for $0.285 \mu\text{mol}/\text{cm}^2$). The absorption model was embedded in the whole body PBTK model to describe the distribution and excretion of chlordecone in young and adult rats. The distribution of chlordecone from blood to various tissue compartments was described. The PBTK model took into account chlordecone binding to albumin and lipoproteins in the blood, preferential uptake by the liver, and the predominant fecal excretion pathway for chlordecone. Once optimized using the experimental data for chlordecone, the PBTK model was used to predict partition coefficients and excretion rates. Tissue concentrations at varying dose levels

were reasonably well estimated if the nonlinear dermal absorption at high doses and the nonlinear uptake of bound chlordecone into the liver were considered.

el-Masri et al. (1995) utilized PBTK and toxicodynamic (TD) modeling to evaluate the toxic interaction between chlordecone and carbon tetrachloride. Chlordecone significantly potentiates the hepatotoxicity and lethality of carbon tetrachloride by interfering with the regeneration process in the liver (see Section 4.4.2). A PBTK model for carbon tetrachloride was adapted and verified using experimental data. The PBTK model was then linked with a TD model based on the mechanistic data for the interaction between chlordecone and carbon tetrachloride in liver cells. The combined model yielded a time course simulation of mitotic, injured, and pyknotic cells following treatment with carbon tetrachloride alone or in combination with chlordecone. The PBTK/TD model was coupled with Monte Carlo simulation techniques to predict the acute lethality of carbon tetrachloride under various exposure conditions. Predictions of lethality were in agreement with experimentally derived values except at very high doses where neurotoxicity led to significant mortality.

4. HAZARD IDENTIFICATION

4.1. STUDIES IN HUMANS—EPIDEMIOLOGY, CASE REPORTS, CLINICAL CONTROLS

Information regarding the health effects of chlordecone in humans comes from studies of a single group of 133 men exposed occupationally to chlordecone at a facility in Hopewell, Virginia (Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). Of the 133 men, 76 experienced neurological symptoms, especially nervousness, headaches, and tremors, sometimes persisting as long as 9–10 months after cessation of exposure (Cannon et al., 1978). In addition, some of the men experienced oligospermia. Sperm count and motility had returned to normal by 5 to 7 years following the cessation of chlordecone exposure and treatment with cholestyramine to reduce chlordecone blood levels (Taylor, 1982). Some workers exposed to high levels of chlordecone developed skin rashes, enlarged livers, and joint pain. Liver enlargement developed in 20 out of 32 workers with high blood levels of chlordecone ($>0.6 \mu\text{g/mL}$) after an average duration period of 5–6 months, although evidence of significant liver toxicity was not found (Guzelian, 1982a; Guzelian et al., 1980; Taylor et al., 1978). Normal results were obtained in all patients for serum bilirubin, albumin, globulin, prothrombin time, cholesterol, alanine aminotransferase (ALT), aspartate aminotransferase (AST), and γ -glutamyl transpeptidase (GGT), and serum alkaline phosphatase (ALP) was only minimally elevated in seven patients. Sulfobromophthalein retention, a measure of liver clearance, was normal in a subset of 18 workers tested (Guzelian et al., 1980). Liver biopsy samples taken from 12 workers with hepatomegaly showed histological changes in the liver that were characterized as nonadverse in nature. These included proliferation of the smooth endoplasmic reticulum (SER) and cytoplasmic accumulation of lipofuscin. No evidence of liver neoplasia, fibrosis, cholestasis, or hepatocellular necrosis was found. Neurological symptoms were reported in workers exposed to high doses of chlordecone for a period of months to years (Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). These symptoms included tremor, headache, irritability, poor recent memory, rapid random eye movements, muscle weakness, gait ataxia, incoordination, and slurred speech. The effects persisted for as long as 9–10 months after cessation of exposure and the start of treatment (Cannon et al., 1978). Martinez et al. (1978) reported that nerve conduction velocity tests, electroencephalography, radioisotope brain scans, computerized tomography, and analyses of cerebral spinal fluid content from these workers were all normal. Sural nerve and skeletal muscle biopsies in workers with detectable neurological impairment exhibited a reduction in the number of unmyelinated axons and a disruption in Schwann cell metabolism (Martinez et al., 1978).

The factory did not follow good industrial hygiene practices. Substantial inhalation, dermal, and oral exposures could have occurred to the workers (Guzelian, 1982a; Guzelian et al., 1980; Cannon et al., 1978). Because of uncertainties regarding exposure routes and levels at the facility and concomitant exposure to the precursors used to manufacture chlordecone, no-observed-adverse-effect levels (NOAELs) or lowest-observed-adverse-effect levels (LOAELs) could not be established for the adverse effects observed on the nervous systems, livers, and reproductive systems of these men. Liver biopsy samples taken from 12 workers with hepatomegaly resulting from intermediate- or chronic-duration exposures to high levels of chlordecone showed no evidence of significant liver toxicity or cancer (Guzelian et al., 1980); however, conclusions from this study are limited by the very small number of workers sampled, uncertainties concerning exposure dose and route, the relatively brief duration of exposures, and the absence of a sufficient latency period for tumor development. The average exposure of the subjects was 5–6 months, and they were examined immediately after exposure (Cannon et al., 1978). A review of biological and epidemiological evidence of cancer found no population-based studies on cancer in humans related to chlordecone exposure (Ahlborg et al., 1995). These case reports of occupationally exposed workers at the pesticide plant (who were repeatedly exposed to high but unmeasured levels for less-than-lifetime durations) indicate that primary target organs for chlordecone toxicity in humans are the nervous system, reproductive organs, skin, and liver.

4.2. SUBCHRONIC AND CHRONIC STUDIES AND CANCER BIOASSAYS IN ANIMALS—ORAL AND INHALATION

Animal studies show effects similar to those reported in occupationally exposed humans including neurological effects, oligospermia, hepatomegaly, and skin rashes, as well as kidney lesions (which were not reported in occupational studies). Chlordecone is moderately lethal by single exposures; oral median lethal dose (LD₅₀) values range from 71 mg/kg-bw for rabbits to 250 mg/kg-bw for dogs (Larson et al., 1979a). The oral LD₅₀ value for rats is 125 mg/kg-bw (Gaines, 1969). In experimental animals, the effects of chlordecone following short-term exposures generally include nervous system effects (tremor and hyperexcitability), liver hypertrophy (with induction of mixed-function oxidases), and structural and ultrastructural changes in the liver, thyroid, adrenal glands, and testes (ATSDR, 1995; U.S. EPA, 1986c; WHO, 1984). In subchronic studies with experimental animals, chlordecone produced tremors and other neurological symptoms, liver hypertrophy with induction of mixed function oxidases, hepatobiliary dysfunction, and centrilobular hepatocellular necrosis. Chlordecone also interferes with reproductive processes in both males (oligospermia) and females (disruption of estrous cyclicity), and it is fetotoxic in experimental animals. Chronic testing of chlordecone in laboratory animals is limited to two studies: NCI (1976a) and Larson et al. (1979a). In a dietary cancer bioassay with chlordecone, NCI (1976a) found a statistically significant increase in the

incidence of and a reduction in the time to detection of hepatic tumors among male (marginal) and female Osborne-Mendel rats and male and female B6C3F₁ mice. The Larson et al. (1979a) study also reported hepatic proliferative lesions, but the determination of whether these represented tumors was equivocal. No data are available concerning the toxicity of chlordecone in animals following inhalation exposure. Studies demonstrating adverse effects in experimental animals following oral exposures are presented below. No studies were available for inhalation or dermal routes of exposure.

4.2.1. Subchronic Studies

4.2.1.1. Oral Exposure Studies

Huang et al. (1980) administered chlordecone by gavage to adult male ICR mice (15/dose group) at 0 (corn oil vehicle), 10, 25, or 50 mg/kg-day. Animals were gavaged daily for up to 24 days. Tremor and hyperexcitability were observed in all mice receiving chlordecone and time to onset was dose-dependent. Loss of body weight (accompanied by reduced food and water consumption) was also apparent in chlordecone-exposed animals, with the greatest loss of body weight coincident with the onset of tremor. The authors speculated that the reduction in body weight among treated mice was due to paralysis and loss of motor control, which resulted in a decreased ability to eat. Upon termination of chlordecone administration, a diminution of tremor and corresponding recovery of body weight were observed in surviving animals. Chlordecone-treated mice showed a high degree of mortality. Time to onset of first death and slope of the mortality curves (cumulative mortality per day) were dose-dependent. Mortality among mice exposed to 50 mg/kg-day began on day 4, and all were dead by day 6. For the 25 mg/kg-day group, mortality began on day 6 and reached 100% by day 11. For the 10 mg/kg-day group, mortality began on day 12 and reached nearly 90% by day 24 of treatment. The control group had no deaths. The cumulative oral LD₅₀ was estimated by the authors to be between 180 and 200 mg/kg. In a follow-up study, Fujimori et al. (1982b) administered chlordecone by gavage to male ICR mice at 0 (corn oil vehicle), 10, 25, or 50 mg/kg-day for 9 consecutive days. The results of this study also demonstrated a dose-response in the time to onset of chlordecone impairment of motor coordination (days 2, 4, and 9 for 50, 25, and 10 mg/kg-day dose groups). The authors examined dopamine, serotonin, and norepinephrine levels in the brain. Significant decreases in whole brain and striatal dopamine levels were seen in animals exhibiting tremors, indicating the dopaminergic pathway may be involved in chlordecone-induced tremor and neurotoxicity.

4.2.1.2. Inhalation Exposure Studies

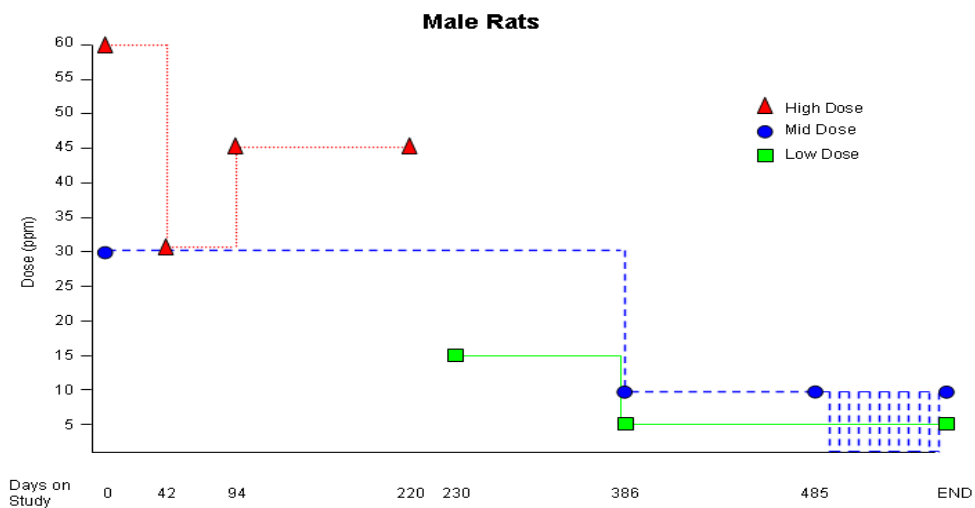
No inhalation exposure studies were found in the literature.

4.2.2. Chronic Studies

4.2.2.1. *Oral Exposure Studies*

Chu et al. (1981a) fed male Sprague-Dawley rats (10/group) diets containing 0 or 1 ppm of chlordecone (0 or 0.07 mg/kg-day reported by the authors) for 21 months. Corn oil was used to dissolve the chemicals, and control diets contained 4% corn oil. Survival and weight gain were similar in treated and control rats. Hematology and clinical chemistry were also unaffected by treatment. Histopathological findings included increases in the incidence of lesions in the liver (5/6 [83%] vs. 3/7 [43%] in controls) and thyroid (4/6 [67%] vs. 1/7 [14%] in controls). The differences in incidences were not statistically significant (Fisher's exact test conducted for this review), although the power of the statistical test to detect a difference at such small sample sizes is low. The lesion in the liver was described as pericentral cytoplasmic vacuolation with mild anisokaryosis, while the thyroid lesion was described as a mild degenerative and proliferative change in the epithelium. The authors reported that the severity of both lesions was increased in chlordecone-treated rats in comparison with controls, although the nature and extent of these differences were not described.

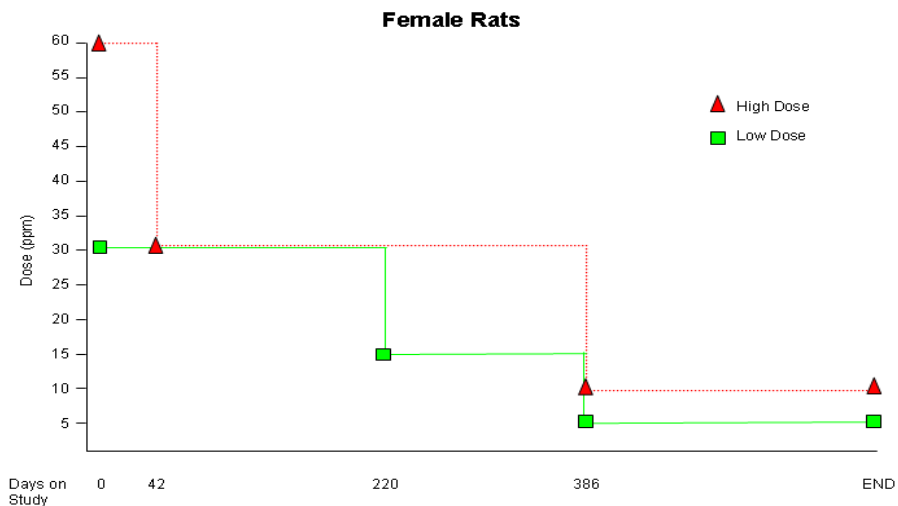
Osborne-Mendel rats and B6C3F₁ mice were exposed to technical-grade chlordecone in the diet for 80 weeks in a study by the National Cancer Institute (NCI, 1976a, b). The test material was reported to contain no more than 2% impurities other than water. Chlordecone was added to finely ground rat chow in acetone (to aid uniform dispersion of the chemical); the diets were mixed for homogeneity and to allow the acetone to evaporate. Corn oil (2%) was added to the diet as a dust suppressant. Dietary concentrations of chlordecone began at 0, 15, 30, or 60 ppm for male rats and 0, 30, or 60 ppm for female rats. Treatment groups comprised 50 rats/sex; however, only 10 animals/sex were used in matched control groups. Pooled control groups (from the same laboratory with birth dates within 3–4 months of the animals in the matched control and exposed groups) contained 105 male rats and 100 female rats. Overt clinical signs of toxicity observed in the treated animals indicated that the initial doses exceeded the maximum tolerated dose in the high exposure groups; consequently, concentrations of chlordecone in feed were reduced (to one-third to one-sixth of the original concentration) during the experiment (after durations ranging from as short as 42 days in high-dose female rats to as long as 386 days in high-dose male rats). The specific dosing regimens for male and female rats are illustrated in Figures 4-1 and 4-2.



*Lines represent changes in the dose levels made throughout the study period. The undulating line for the mid dose from day 485 until the end of the study represents a recovery period of a week between doses for the last 75 days of the study. Additionally, the low dose group was added in the middle of the study period.

Source: NCI (1976a, b).

Figure 4-1. Dosing regimen for male rats.



*Lines represent changes in the dose levels made throughout the study period.

Source: NCI (1976b).

Figure 4-2. Dosing regimen for female rats.

The initial group of high-dose male rats was discontinued due to excess toxicity; however, nine rats were transferred to the lower dose group in the study. A new dose group of male rats was started 8 months after the beginning of the study. Time-weighted-average dietary concentrations were reported by the authors to be 0, 8, or 24 ppm for male rats and 0, 18, or 26 ppm for female rats. Doses estimated from U.S. EPA (1988) reference values for body weight and food consumption were calculated¹: 0, 0.6, or 1.7 mg/kg-day for male rats and 0, 1.4, or 2.0 mg/kg-day for female rats. Following the 80-week exposure, surviving rats were sacrificed at 112 weeks. The following tissues were taken from sacrificed animals, and those dying early, for histological examination: brain, pituitary, lymph nodes, thyroid, parathyroid, salivary glands, lung, heart, diaphragm, stomach, duodenum, jejunum or ileum, large intestine, pancreas, adrenal glands, kidney, liver, skin, gonads, bladder, prostate or uterus, and femur with marrow.

Clinical signs of chlordecone toxicity, including tremor and dermatological changes, were indicated in the NCI (1976a) report, although incidence by dose was not reported (NCI, 1976a, b). Survival was reduced for high-dose male and female rats (NCI, 1976a). Percentages of male rats surviving to study termination (112 weeks) were 63% for pooled controls, 90% for matched controls, 60% for the low-dose group, and 42% for the high-dose group; for female rats, the respective percentages were 61, 70, 56, and 40%. The decreases in survival occurred primarily during the second year of the study, although some early mortality was observed among high-dose male rats (4 animals in the first 4 months). Many of the treated rats also showed decreases in food consumption and body weight gain (NCI, 1976b). In male rats, body weight gain at 79 weeks was 82 and 79% of controls for the low- and high-dose groups, respectively. Body weight gain in female rats at 79 weeks was 76 and 66% of control for the low- and high-dose groups, respectively. Some high-dose males were observed to have bleeding of the eyes and nose during the first 4 months of the study, and, by week 5 of the study, most high-dose females showed generalized tremors. By week 28, many low-dose females were also experiencing tremors. The incidence of tremors and other clinical signs (rough hair coat, dermatitis, anemia) was low to moderate during the remainder of the first year, but gradually increased during the second year of the study. The authors reported that rats surviving to study termination were generally in very poor physical condition, though more specific data regarding occurrence of clinical signs were not reported.

In rats, the incidence of noncancer lesions were reported in summary tables included in unpublished raw data for the bioassay (NCI, 1976b). These tables showed chronic kidney inflammation in low-dose male rats and high-dose female rats but did not confirm the presence of extensive noncancer liver lesions in male or female rats. Liver tumors described as

¹Calculation: mg/kg-day = (ppm in feed × kg food/day)/kg-bw. Reference food consumption rates of 0.036 kg/day (males) and 0.030 kg/day (females) and reference body weights of 0.514 kg (males) and 0.389 kg (females) were used (U.S. EPA, 1988); doses calculated are for dosing period and do not reflect the period between the end of dosing (80 weeks) and terminal sacrifice (112 weeks for rats and 90 weeks for mice).

hepatocellular carcinomas were observed in high-dose female rats at an incidence that was significantly elevated compared with the pooled control incidence (0/100, 0/10, 1/49, and 10/45 in the pooled control, matched control, low-dose, and high-dose groups, respectively). Incidences of male rats with hepatocellular carcinomas were 0/105, 0/10, 1/50, and 3/44, respectively. The incidence of carcinomas in high-dose males was significant ($p = 0.049$) in comparison with pooled controls. The incidence of hepatocellular carcinomas was not statistically significant in comparison with matched controls for rats of either sex. A significant dose-response trend was observed for the incidence of hepatocellular carcinoma in both male and female rats (Cochran-Armitage test conducted for this review). Hepatocellular carcinomas were described as large, poorly circumscribed masses that were well differentiated without vascular invasion or metastases. Liver tumors described as neoplastic nodules were also observed, but not at elevated incidences in exposed groups compared with control groups. Neoplastic nodule incidences were reported to be 0/10, 2/50, and 0/44 in the matched control, low-dose, and high-dose male rats and 1/10, 0/49, and 2/45 in the corresponding groups of female rats. The incidence and time to tumor data for hepatocellular carcinoma in rats in the NCI (1976a) report are summarized in Table 4-1.

Table 4-1. Incidence and time to tumor of hepatocellular carcinoma in rats

| Osborne-Mendel rats | Exposure group | | | |
|--|-----------------|----------------|-----------|---------------------|
| | Matched control | Pooled control | Low dose | High dose |
| Male (0, 0.6, or 1.7 mg/kg-day) ^a | 0/10 | 0/105 | 1/50 | 3/44 ^{b,c} |
| Time to first tumor (weeks) | NA ^d | NA | 112 weeks | 108 weeks |
| Female (0, 1.4, or 2.0 mg/kg-day) ^a | 0/10 | 0/100 | 1/49 | 10/45 ^c |
| Time to first tumor (weeks) | NA | NA | 87 weeks | 83 weeks |

^aDoses were calculated for this review using the allometric equation for food consumption by laboratory animals with time-weighted concentrations from NCI (1976a) and reference body weights from U.S. EPA (1988).

^bMarginal increase ($p = 0.049$) compared with pooled controls.

^cStatistically significant increase in incidence as compared with pooled controls, using one-tail ($p < 0.05$) Fisher's exact test for 2×2 contingency table (NCI, 1976a).

^dNA = not applicable.

Source: NCI (1976a).

In addition to the liver, the rats developed tumors in other organs of the endocrine system (NCI, 1976a). Table 4-2 shows the incidence of these tumors by organ and tumor type. The incidence rate for all tumor types combined for each of these systems (endocrine or reproductive) was not statistically increased as compared with controls (Fisher's exact test conducted for this review). Individual tumor types were also not significantly increased, and no dose-response trend was observed (Cochran-Armitage test conducted for this review).

Table 4-2. Summary of endocrine and reproductive system tumor incidence among rats exposed to chlordecone

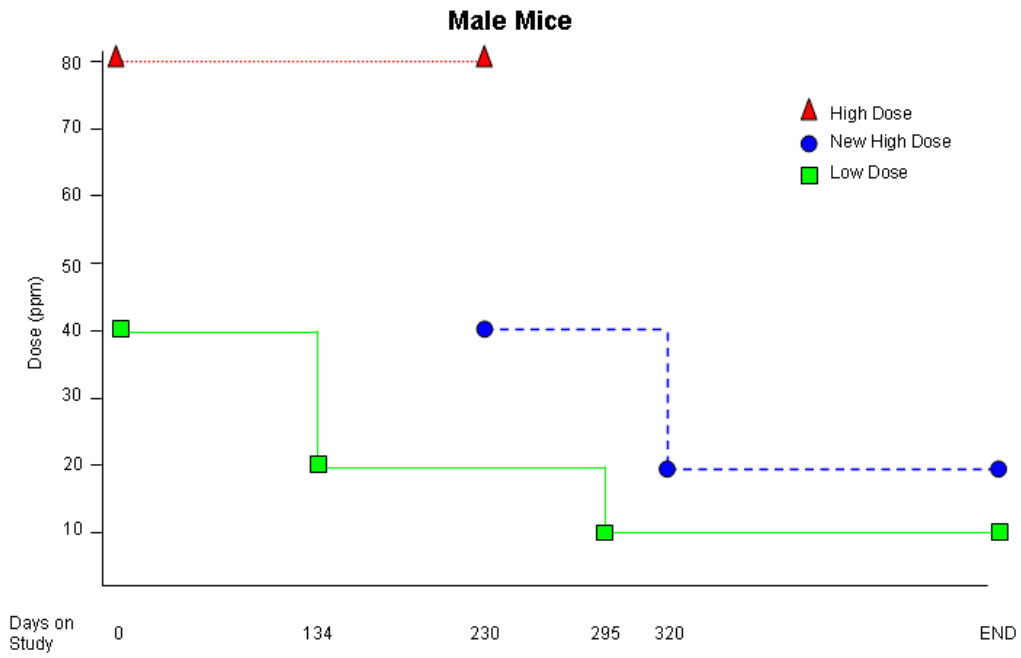
| | | Males | | | Females | | |
|--|--------------------------------------|---------|---------------|---------------|---------|---------------|---------------|
| | | Control | 0.6 mg/kg-day | 1.7 mg/kg-day | Control | 1.4 mg/kg-day | 2.0 mg/kg-day |
| Number of rats | | 10 | 50 | 44 | 10 | 49 | 45 |
| Number of rats with any type of tumor ^a | | 3 (30%) | 24 (48%) | 16 (36%) | 7 (70%) | 29 (59%) | 31 (69%) |
| Endocrine Organs | Pituitary chromophobe adenoma | 2 (20%) | 12 (24%) | 5 (11%) | 3 (30%) | 13 (26%) | 4 (9%) |
| | Pituitary adenocarcinoma | – | – | 1 (2%) | – | – | – |
| | Thyroid follicular-cell carcinoma | – | 3 (6%) | – | – | – | 1 (2%) |
| | Thyroid follicular-cell adenoma | – | 2 (4%) | – | – | 1 (2%) | – |
| | Thyroid C-cell adenoma | – | 3 (6%) | – | – | 2 (4%) | 1 (2%) |
| | Thyroid C-cell carcinoma | – | 1 (2%) | – | – | – | 1 (2%) |
| | Parathyroid adenoma | – | – | 1 (2%) | – | – | – |
| | Pancreatic islet cell adenoma | – | 1 (2%) | 1 (2%) | – | 1 (2%) | – |
| | Adrenal cortical adenoma | – | 1 (2%) | – | – | – | 2 (4%) |
| Mammary Gland | Mammary gland fibroadenoma | – | 1 (2%) | 1 (2%) | 4 (40%) | 4 (8%) | 1 (2%) |
| | Mammary gland adenoma | – | – | 1 (2%) | 2 (20%) | 1 (2%) | – |
| | Mammary gland fibroma | 1 (10%) | – | – | – | – | – |
| | Mammary gland adenocarcinoma | – | – | – | – | 2 (4%) | – |
| | Mammary gland fibrolipoma | – | – | – | – | – | 1 (2%) |
| Reproductive Organs | Uterus endometrial/stromal polyp | – | – | – | – | 3 (6%) | 1 (2%) |
| | Uterus malignant lymphoma | – | – | – | – | 1 (2%) | – |
| | Uterus squamous cell carcinoma | – | – | – | – | – | – |
| | Ovary arrhenoblastoma | – | – | – | – | 1 (2%) | – |
| | Ovary granulosa-cell tumor | – | – | – | – | – | 1 (2%) |
| | Cervix uteri squamous cell carcinoma | – | – | – | – | 1 (2%) | – |

^aSome animals had multiple tumors.

Source: NCI (1976a).

In mice, dietary concentrations of chlordecone began at 0 or 40 ppm (two groups at this concentration) for males and 0, 40, or 80 ppm for females. Treatment groups comprised 50 mice/sex; however, only 10 female mice and 19 male mice were used as matched controls. Pooled control groups (from the same laboratory with birth dates within 3–4 months of the animals in the matched control and exposed groups) contained 49 male mice and 40 female mice. Overt clinical signs of toxicity observed in the high-dose male and female mice indicated that the maximum tolerated dose was exceeded in those exposure groups; consequently, concentrations of chlordecone in feed for all dose groups were reduced (one-fourth to one-half of the original concentration) during the experiment. The specific dosing regimens for male and female mice were described in detail in unpublished raw data for the bioassay (NCI, 1976b) and are illustrated in Figures 4-3 and 4-4. The initial high-dose group of male mice was discontinued due to excess toxicity, and a new group was started 7 months later after the beginning of the study. Time-weighted-average dietary concentrations were reported by the authors to be 0, 20, or 23 ppm for male mice and 0, 20, or 40 ppm for female mice. Doses estimated from U.S. EPA (1988) reference values for body weight and food consumption were calculated²: 0, 3.4, or 3.9 mg/kg-day for male mice and 0, 3.5, or 7.0 mg/kg-day for female mice. Following the 80-week exposure, surviving mice were sacrificed at 90 weeks. Histological examination was similar to that described previously for rats.

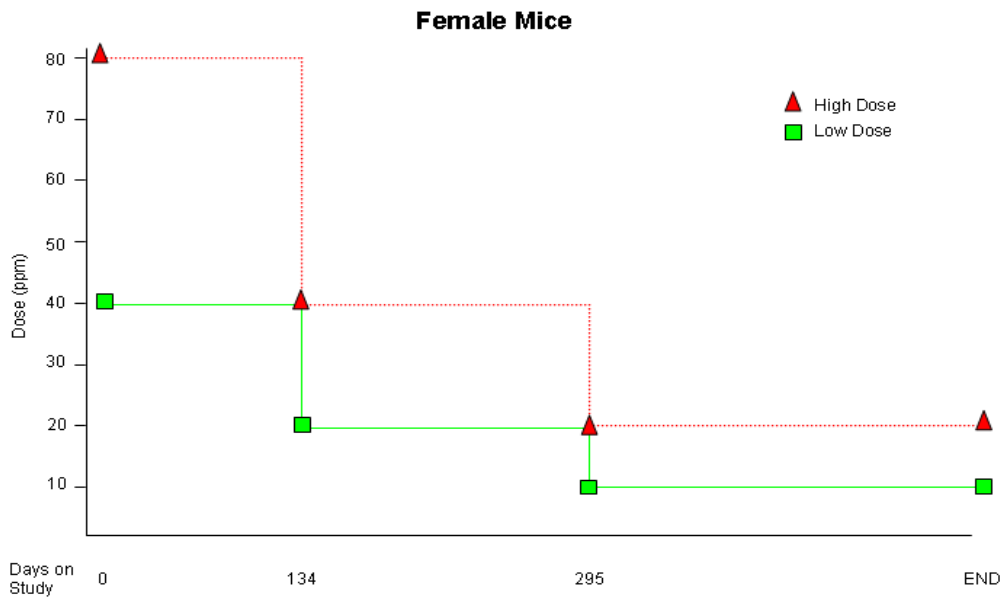
²Calculation: mg/kg-day = (ppm in feed × kg food/day)/kg-bw. Reference food consumption rates of 0.0064 kg/day (males) and 0.0061 kg/day (females) and reference body weights of 0.0373 (males) and 0.0353 kg (females) were used (U.S. EPA, 1988).



*Lines represent changes in the dose levels made throughout the study period.

Source: NCI (1976a, b).

Figure 4-3. Dosing regimen for male mice.



*Lines represent changes in the dose levels made throughout the study period.

Source: NCI (1976a, b).

Figure 4-4. Dosing regimen for female mice.

Survival was reduced for male mice at both the high and low dose, though survival rates in female mice at both dose levels were comparable with those of controls (NCI, 1976a). The percentages of male mice surviving to study termination at 90 weeks were 92% for pooled controls, 90% for matched controls, 58% for the low-dose group, and 50% for the high-dose group. The percentages of survival for female mice were 85% for pooled controls, 90 for matched controls, 84% for the low-dose group, and 84% for the high-dose group. The decreases in survival occurred primarily during the second year of the study, although some early mortality was observed. Decreases in food consumption and body weight gain were less pronounced in mice as compared to rats (NCI, 1976b). In male mice, body weight gain at 81 weeks was 93 and 88% of control for the low- and high-dose groups, respectively. Body weight gain in female mice at 81 weeks was 94% and 88% of control for the low- and high-dose groups, respectively. A comparison of survival rates and body weight gain for animals in the NCI study is presented in Table 4-3.

Table 4-3. Percent body weight gain and percent survival of chlordecone-exposed rats and mice

| | Time-weighted-average daily dose (mg/kg-day) | Survival (%) | Body weight gain (%) | Liver tumor incidence (%) | Time to 1 st tumor (weeks) |
|-------------|--|--------------|----------------------|---------------------------|---------------------------------------|
| Male rats | 0 (room controls) | 63 | | 0/105 | NA ^a |
| | 0 (matched controls) | 90 | | 0/10 | NA |
| | 0.6 | 60 | 82 | 1/50 (2) | 112 |
| | 1.7 | 42 | 79 | 3/44 (7) ^b | 108 |
| Female rats | 0 (room controls) | 61 | | 0/100 | NA |
| | 0 (matched controls) | 70 | | 0/10 | NA |
| | 1.4 | 56 | 76 | 1/49 (2) | 87 |
| | 2.0 | 40 | 66 | 10/45 (22) ^b | 83 |
| Male mice | 0 (room controls) | 92 | | 8/49 (16) | 87 |
| | 0 (matched controls) | 90 | | 6/19 (31) | 87 |
| | 3.4 | 58 | 93 | 39/48 (81) ^b | 70 |
| | 3.9 | 50 | 88 | 43/49 (88) ^b | 62 |
| Female mice | 0 (room controls) | 85 | | 0/40 | NA |
| | 0 (matched controls) | 90 | | 0/10 | NA |
| | 3.5 | 84 | 94 | 26/50 (52) ^b | 87 |
| | 7.0 | 84 | 88 | 23/49 (47) ^b | 76 |

^aNA = not available.

^bStatistically significant increase in incidence as compared with matched or pooled controls, using one-tail Fisher's exact test ($p < 0.05$).

Source: NCI (1976a).

Clinical signs of chlordecone toxicity were reported in mice; however, the incidence by dose was not reported (NCI, 1976a, b). High-dose female mice developed tremors during the first week of the study that persisted to study termination. Tremors were also observed in some high-dose male mice, and about 20% of high-dose males were highly excitable during the second year of the study. Abdominal distention was first observed in high-dose males at week 45 and high-dose females at week 68, presumably associated with hepatic hypertrophy. Palpable abdominal masses were found in high- and low-dose males during the second year of the study. Alopecia, rough hair coats, and tail sores were seen primarily in males and were thought to be due to fighting. More specific data regarding occurrence of clinical signs were not reported.

In mice, statistically significant elevated incidences of hepatocellular carcinomas were found in both exposed groups compared with matched and pooled control incidences (NCI, 1976a). Incidences for matched control, low-, and high-dose groups were 6/19, 39/48, and 43/49 for male mice and 0/10, 26/50, and 23/49 for female mice. The incidence in control male mice was reported as abnormally high. Two of the pooled control male mice had hepatocellular carcinomas. Combining the matched and pooled control male mouse groups resulted in an overall incidence of 8/49 for control male mice. Hepatocellular carcinomas in mice were described as varying from demarcated nodules to large masses that were well differentiated without vascular invasion or metastases. Extensive liver hyperplasia also was found in both sexes in both low- and high-dose mouse groups. Incidences for liver hyperplasia were not specified, but the report noted that “a few matched controls of each sex also had liver hyperplasia although the incidence was quite low as compared to the treated groups.” No tumors of other endocrine organs were reported, aside from one ovary cystadenoma in a single high-dose female (1/49 or 2% incidence rate). No elevated incidences of tumors at other tissue sites were found in exposed mice compared with controls. The incidence and time-to-tumor data for hepatocellular carcinoma in the NCI (1976a) report are summarized in Table 4-4. No exposure-related noncancer lesions were mentioned other than the liver atypia and nodular and diffuse hyperplasia (NCI, 1976a, b). Induction of noncancerous liver lesions (i.e., hyperplasia) was observed at all dose levels for each sex and species. Thus, freestanding LOAELs identified for this study are 0.6, 1.4, 3.4, and 3.5 mg/kg-day for male rats, female rats, male mice, and female mice, respectively.

Table 4-4. Incidence and time to tumor of hepatocellular carcinoma in mice

| Mouse/B6C3F ₁ | Exposure group | | | |
|--|-----------------|----------------|--------------------------|--------------------------|
| | Matched control | Pooled control | Low-dose group | High-dose group |
| Male (0, 3.4, or 3.9 mg/kg-day) ^a | 6/19 (31%) | 8/49 (16%) | 39/48 ^b (81%) | 43/49 ^b (88%) |
| Time to first tumor (weeks) | 87 weeks | 87 weeks | 70 weeks | 62 weeks |
| Female (0, 3.5, or 7.0 mg/kg-day) ^a | 0/10 (0%) | 0/40 (0%) | 26/50 ^b (52%) | 23/49 ^b (47%) |
| Time to first tumor (weeks) | NA ^c | NA | 87 weeks | 76 weeks |

^aDoses were calculated for this review using the allometric equation for food consumption by laboratory animals with time-weighted concentrations from NCI (1976a) and reference body weights from U.S. EPA (1988).

^bStatistically significant increase in incidence as compared to matched or pooled controls, using one-tail ($p < 0.05$) Fisher's exact test for 2×2 contingency table (NCI, 1976a).

^cNA = not applicable.

Source: NCI (1976a).

The NCI (1976a) study provides evidence of carcinogenicity in Osborne-Mendel rats and B6C3F₁ mice; however, decreases in survival rates and decreased body weight gain indicate that excessively high doses were utilized in all animal groups except the low- and high-dose female mice (see Table 4-4).

In another chronic study, groups of 40 male and 40 female Wistar rats were fed diets containing 0, 5, 10, 25, 50, or 80 ppm of chlordecone for up to 2 years (Larson et al., 1979a). Larson et al. (1979a) added chlordecone to warmed corn oil before combining it with the food. From food consumption and body weight data graphically presented in Larson et al. (1979a) for 5–8 time points measured throughout the study, time-weighted-average food consumption rates were estimated for the 5 through 80 ppm groups as 49, 53, 59, 73, and 80 g food/kg-bw-day for males and 56, 55, 69, 83, and 93 g food/kg-bw-day for females. Using average food consumption rates and averaged body weights (between males and females), doses were estimated to be 0, 0.3, 0.5, 1.6, 3.9, and 7.0 mg/kg-day. In a separate phase of the experiment, groups of 40 males and 40 females were exposed to 0 or 1 ppm for up to 2 years. Because food consumption data were not reported for the 1 ppm group, an estimated dose of 0.06 mg/kg-day was calculated by assuming food consumption equal to the 5 ppm group. Groups of five rats/sex/dose were sacrificed at 3 and 12 months. Another 3–5 rats/sex/group were sacrificed after 12 months of exposure and a 4-week recovery period. Remaining rats were sacrificed at 24 months. Because of serial sacrifices and early mortality in the high-dose groups, effective numbers of animals available for histological examination at the conclusion of the study were greatly reduced with only four animals/group in the highest dose group of male and female rats. From samples collected at 3-month intervals, hematocrit, hemoglobin, and total and differential white cell counts were measured in blood, and reducing substances and protein were measured in

urine. Additional blood studies were performed at 3 months for platelet count, prothrombin clotting time, and serum calcium. Oxygen consumption was measured by spirometry at 9 months. Organ-to-body-weight ratios (liver, kidneys, heart, spleen, and testes) were determined in sacrificed rats. The following tissues were taken from sacrificed rats for histopathological study: brain, spinal cord, heart, lung, liver, kidney, spleen, gut, urinary bladder, bone marrow, skeletal muscle, skin, pancreas, thyroid, adrenal, pituitary, and gonad.

Tremors developed in the 3.9 and 7.0 mg/kg-day groups within a few weeks of the start of the study and became progressively more severe with time (Larson et al., 1979a). Slight tremors were noted in some rats at 1.6 mg/kg-day after 3 months, becoming moderate in severity after 5–6 months, but then regressing. Tremors were not observed at ≤ 0.5 mg/kg-day. The incidence of tremors was not reported. All rats in the 3.9 and 7.0 mg/kg-day groups died during the first 6 months. Long-term survival was reduced in the 1.6 mg/kg-day females (measured at 1 and 2 years, data not shown). Body weights were depressed after 3 weeks of study in males at ≥ 1.6 mg/kg-day and in females at ≥ 0.3 mg/kg-day. Food consumption (per body weight) tended to increase with concentration of chlordecone in the feed. Metabolic rate (measured by oxygen consumption) increased with dose in both males and females, although statistical significance was achieved only in males at 1.6 mg/kg-day (the highest dose with survivors remaining when tested at 9 months). Hematology analyses revealed no differences related to treatment. Increases in urinary protein concentrations or proteinuria (a clinical indicator of glomerular dysfunction) were reported in both male and female rats exposed to ≥ 0.3 mg/kg-day for 6–24 months, though statistical analysis was not performed on these data because of incomplete data reporting. Proteinuria was not observed in rats exposed to 0.06 mg/kg-day in a separate phase of the experiment (the time of analysis and other details were not reported). Relative liver weight increased with dose at 3, 12, and 24 months in both male and female rats. The difference from controls was statistically significant at ≥ 1.6 mg/kg-day in males and ≥ 0.5 mg/kg-day in females. Relative testes weights were significantly decreased in the 3.9 and 7.0 mg/kg-day groups at the 3-month sacrifice. Relative weight changes in the kidneys and other organs were not remarkable. Absolute organ weights were not reported.

Histopathological examination of five rats (randomly selected) from each sex at each feeding level at 13 weeks revealed minimal congestion of the liver at 0.5 mg/kg-day and more degenerative changes in the liver at higher doses (Larson et al., 1979a). There was a trend in dose-response for degenerative liver changes. Swollen liver cells were noted in 4/5 males and 5/5 females in the 3.9 mg/kg-day group and 5/5 males and 3/5 females in the 7.0 mg/kg-day group (compared with 0/10 males and 0/10 females in the control groups). The liver-to-body-weight ratios were significantly increased in the 3.9 and 7.0 mg/kg-day groups for both sexes. Histological examination also uncovered a dose-related increase in the incidence and severity of testicular atrophy at 13 weeks, though not at 1–2 years. The study authors did not speculate as to why testicular atrophy was observed after 13 weeks but not at the chronic time point. Interim (3-

month) gross and histopathologic examinations performed on 10 control males and 5 chlordecone-treated males/group revealed statistically significantly increased incidences of chlordecone-induced testicular atrophy (Table 4-5). The atrophy was described as minimal in the controls and generally increased in severity with increasing chlordecone concentration. Also, the testes-to-body-weight ratios in males were significantly decreased in the 3.9 and 7.0 mg/kg-day groups. The study identified a NOAEL of 0.5 mg/kg-day and a LOAEL of 1.6 mg/kg-day for testicular atrophy in male rats exposed to chlordecone in the diet for 13 weeks.

Table 4-5. Testicular atrophy in male rats receiving chlordecone in the diet for 3 months

| Dietary level (ppm) | 0 | 5 | 10 | 25 | 50 | 80 |
|--|------|-----|-----|------------------|------------------|------------------|
| Average dose ^a (mg/kg-day) | 0 | 0.3 | 0.5 | 1.6 | 3.9 | 7.0 |
| Incidence of testicular atrophy ^b | 1/10 | 0/5 | 1/5 | 4/5 ^c | 4/5 ^c | 5/5 ^c |

^aAverage dose to rats, based on graphically depicted food consumption data presented by the authors.

^bStatistically significant dose-response trend according to the Cochran-Armitage trend test ($p < 0.01$) performed for this review.

^cStatistically significantly different from controls according to Fisher's exact test ($p < 0.05$) performed for this review.

Source: Larson et al. (1979a).

At the 12-month sacrifice, congestion of the liver was reported for treated groups, but details were not reported. No treatment-related lesions were observed after 12 months of treatment and a 4-week recovery period.

Histopathological examination of rats sacrificed after 2 years and rats that died during the second year showed exposure-related lesions only in the liver and kidney (Larson et al., 1979a). Incidence data for liver and kidney effects are presented in Table 4-6. The principal renal lesion was glomerulosclerosis, or scarring of the system of capillaries that comprise the glomeruli. The increased incidence of glomerulosclerosis was statistically significant (Fisher's exact test performed for this review) in the 0.3, 0.5, and 1.6 mg/kg-day females compared with controls. The background incidence of glomerulosclerosis in male rats was high (56% as compared to 12% in female rats) and, as such, male rat incidence data for glomerulosclerosis did not achieve statistical significance. Incidences of liver lesions (predominately fatty changes and hyperplasia) in male and female rats were also statistically increased by chlordecone administration. The hepatic lesions in three females in the 0.5 mg/kg-day group and one female and two males in the 1.6 mg/kg-day group were described by the authors as being possibly "carcinomatous in nature"; however, the authors reported that an independent review by four pathologists found the evidence for carcinogenic responses in this study to be equivocal.

Table 4-6. Incidence of histopathologic liver lesions (fatty changes and hyperplasia) and renal glomerulosclerosis in male and female Wistar rats following administration of chlordecone in the diet for 1–2 years

| Endpoint ^a | Dose (mg/kg-day) | | | | |
|---------------------------------|------------------|------|-------------------|-------------------|------------------|
| | 0 | 0.06 | 0.3 | 0.5 | 1.6 |
| Liver lesions ^b | | | | | |
| Male rats | 1/22 | 1/11 | 2/6 | 2/9 | 3/4 ^c |
| Female rats | 2/34 | 1/13 | 2/17 | 4/12 ^c | 1/4 |
| Glomerulosclerosis ^b | | | | | |
| Male rats | 12/22 | 3/11 | 4/6 | 6/9 | 3/4 |
| Female rats | 4/34 | 2/13 | 8/17 ^c | 8/12 ^c | 3/4 ^c |

^aThe number of animals reported relates to the number of animals analyzed between 1 and 2 years. Due to interim measurements, the approximate number of animals/sex/dose group after 12 months is 25.

^bThe dose-response trend was also statistically significant for each data set according to the Cochran-Armitage trend test performed for this review.

^cStatistically different from control groups according to Fisher's exact test ($p < 0.05$) performed for this review.

Source: Larson et al. (1979a).

This study identified 5 ppm (0.3 mg/kg-day) as a LOAEL and 1 ppm (0.06 mg/kg-day) as a NOAEL for kidney effects (proteinuria and increased incidence of glomerulosclerosis) in female rats. Also observed were increased incidences of hepatic lesions; these increases were statistically significant (Fisher's exact test performed for this review) starting at 1.6 mg/kg-day in males and at 0.5 mg/kg-day in females. Higher doses (3.9 and 7.0 mg/kg-day) produced overt clinical signs (tremors) and mortality in the rats.

Larson et al. (1979a) also conducted a long-term study in dogs. Groups of two male and two female purebred beagle dogs were fed diets containing 0, 1, 5, or 25 ppm of chlordecone for up to 128 weeks, beginning at an age of about 6 months. Two dogs in the 25 ppm group were sacrificed at the end of week 124; the remaining dogs were sacrificed during week 128. Organ-to-body weights were determined, and 17 tissues were taken for histopathological examination: brain, spinal cord, heart, lung, liver, kidney, spleen, gut, urinary bladder, bone marrow, skeletal muscle, skin, pancreas, thyroid, adrenal, pituitary, and gonad. The same hematological and urine endpoints as those described for the rat studies were determined in samples collected before exposure and at 3-month intervals during exposure. Using reference body weights and food consumption rates of 10.5 and 0.2 kg dry food/day, respectively, for beagle dogs (U.S. EPA, 1988), doses were estimated to be 0, 0.02, 0.1, and 0.5 mg/kg-day (the authors did not report food consumption data, body weight data, or estimated dose levels for the dogs). Three dogs died during the study, showing severe dermatitis that did not appear to be related to exposure (one control dog during week 71, one 0.02 mg/kg-day dog during week 48, and one 0.1 mg/kg-day dog during week 50). Body weight gain in the 0.5 mg/kg-day group was reported to be lower than the weight gain in the control dogs during the second year of exposure, but the

magnitude of the decrease was not reported and the data were not shown. Decreased food efficiency (kg body weight gain/kg food consumed) was suggested by measurements of food consumption, but again the data were not shown. The only statistically significant changes associated with exposure to chlordecone were a moderate (37%) increase in relative liver weight in dogs from the 0.5 mg/kg-day group (males and females combined) and slight changes (less than about 25%) in relative kidney (increase), heart (increase), and spleen (decrease) weight in the same group. Absolute organ weights were not reported. No exposure-related changes were reported for clinical signs of toxicity; hematological, histopathological, or urinalysis endpoints; sulfobromophthalein retention; or serum cholinesterase. Interpretation of this study is limited by the small number of dogs tested, the deaths of three dogs during the study for reasons not apparently related to treatment, and the reporting of results, including failure of the researchers to present data to support the reported decrease in body weight in dogs from the 0.5 mg/kg-day group during the second year of the study. Nevertheless, the statistically significant changes in organ-to-body-weight ratios support occurrence of an adverse effect on body weight, and the increase in relative liver weight is consistent with other studies demonstrating hepatic toxicity with chlordecone exposure. Therefore, the results of this study suggest a LOAEL of 25 ppm (0.5 mg/kg-day) and a NOAEL of 5 ppm (0.1 mg/kg-day), based on decreased body weight and organ-to-body-weight changes (without histological changes) in beagle dogs fed chlordecone in the diet for up to 128 weeks.

4.3. REPRODUCTIVE/DEVELOPMENTAL STUDIES

4.3.1. Reproductive Toxicity Studies

Information on reproductive effects in humans is restricted to findings of oligospermia, reduced sperm motility, and decreased libido in a group of men who were occupationally exposed to chlordecone for periods up to 1.5 years (Taylor, 1985, 1982; Guzelian, 1982a; Taylor et al., 1978). Sperm concentration and motility had returned to normal upon follow-up 5 years following cessation of chlordecone exposure. Even though two of seven workers sired children, there is no indication of the true denominator of how many were trying to conceive and/or the fertility rate. In one worker, low sperm count persisted (Taylor, 1985). No information is available concerning chlordecone-induced reproductive effects in women.

Reproductive toxicity has been assessed in some animal studies, but not in adequately designed multiple generation studies. Available animal data suggest that chlordecone is a male reproductive toxicant, causing alteration of sperm parameters at low doses and testicular atrophy at higher doses. Persistent vaginal estrus (PVE) is reported to occur in exposed females, and decreased reproductive success has been demonstrated. No animal studies are available to assess the developmental or reproductive toxicity of chlordecone by the inhalation route of exposure.

Huber (1965) performed a series of experiments designed to assess reproductive toxicity in mice exposed to chlordecone in the diet. In a pilot reproduction study (group A), 3-month-old

male and female mice of mixed parentage (eight pairs/group) were administered chlordecone (technical-grade chlordecone, 93.6% purity) in the diet at concentrations of 0, 10, 30, or 37.5 ppm for 1 month prior to mating and during the 100 days following individual pairing within each exposure group. Corresponding chlordecone doses of 0, 1.9, 5.6, and 7.0 mg/kg-day were estimated for males and females combined by using reference values for food consumption and body weight from U.S. EPA (1988). The 100-day treatment period allowed sufficient time for mating pairs to produce two litters. Individual males were housed with individual females except during the period of gestation and weaning of offspring. Reproductive parameters assessed included number of pairs producing first and second litters, average number of young per litter, percent survival of offspring, and the average time required to produce the offspring (expressed as pair days/litter [number of pairs \times 100 days/number of litters produced] and pair days/offspring [number of pairs \times 100 days/number of offspring]). Vaginal smears were taken daily for 3–4 weeks for analysis of the estrous cycle following the termination of the reproduction phase. Smears were taken in one group after assessment of reproduction and in another group prior to mating.

In the chlordecone-treated groups, the number of pairs producing first and second litters, the average number of young/litter, and the percent survival of offspring was lower compared with controls. The average time required to produce offspring during the treatment period was greater in chlordecone-treated pairs than controls. However, except for the quantal data presented for pairs producing litters, the data presented for the continuous parameters (average number of offspring, pair days/litter, and percent survival of offspring) did not include a measure of the variance and thus were not adequate for statistical analysis. Visual evaluation of the data indicate a reduction in reproductive success at doses >5.6 mg/kg-day. Statistical analysis of the number of pairs producing second litters (Fisher's exact test performed for this review) revealed a significant reduction in the 5.6 and 7.0 mg/kg-day exposure groups relative to controls (Table 4-7).

Table 4-7. Effects of dietary chlordecone on reproduction in male and female mice (of mixed parentage) treated for 1 month prior to mating and for 100 days following the initiation of mating

| Dietary level (ppm) | Average dose ^a (mg/kg-day) | Pairs producing first litter | Pairs producing second litter | Average number offspring/litter | Percent survival of offspring | Pair days/litter | Pair days/offspring |
|---------------------|---------------------------------------|------------------------------|-------------------------------|---------------------------------|-------------------------------|------------------|---------------------|
| 0.0 | 0.0 | 7/8 | 5/8 | 7.7 | 89 | 67 | 8.7 |
| 10.0 | 1.9 | 6/8 | 4/8 | 7.1 | 87 | 80 | 11.3 |
| 30.0 | 5.6 | 4/8 | 0/8 ^b | 4.7 | 26 | 200 | 42.1 |
| 37.5 | 7.0 | 3/8 | 0/8 ^b | 4.0 | 42 | 267 | 66.7 |

^aAverage doses to male and female mice (combined), based on reference values for subchronic body weight and food consumption taken from U.S. EPA (1988).

^bStatistically different from control groups according to Fisher's exact test ($p < 0.05$), performed for this review.

Source: Huber (1965).

In another phase (group B) of the study, 4-month-old BALB/cJaxGnMc mice (14 pairs/group) were administered chlordecone in the diet at concentrations of 0 or 40 ppm for 2 months before mating and during a 100-day reproduction period which included mating, gestation, and lactation (Huber, 1965). Otherwise, the study design was the same as that used for group A. The corresponding chlordecone dose was 7.6 mg/kg-day (estimated for males and females combined, using reference values for food consumption and body weight from U.S. EPA [1988]). Following the termination of treatment, a second reproduction phase was performed for 100 days and consisted of crossover matings (control males with control females, control females with chlordecone-treated males, and chlordecone-treated females with control males).

The results are summarized in Table 4-8. During the initial reproduction period, each of the control (0 ppm) pairs produced two litters. No offspring were produced by the pairs of mice treated with 7.6 mg/kg-day of chlordecone. The ability to produce offspring was restored during the post-treatment reproduction period. Results of crossover matings indicated that female mice were slightly more affected by chlordecone than males; however, information concerning the statistical significance of the findings was not provided by the author.

Table 4-8. Effects of dietary chlordecone (0 or 40 ppm) on reproduction in BALB/cJaxGnMc mice during 100 days of treatment (preceded by 2 months of pre-mating treatment) and during 100 days of a crossover-mating period following the termination of treatment

| | Reproduction period during chlordecone treatment | | Crossover reproduction period following termination of chlordecone treatment | | |
|--------------------------|--|---------------|--|-------------------------------------|-------------------------------------|
| | Controls | Treated pairs | Controls | Control male × treated female | Control female × treated male |
| Pairs with first litter | 14/14 | 0/14 | 5/5 | 8/10 | 10/10 |
| Pairs with second litter | 14/14 | 0/14 | 4/5 | 5/10 | 6/10 |
| Offspring/litter | 7.1 | – | 7.2 | 4.5 | 5.6 |
| Offspring survival (%) | 89 | – | 87.3 | 76.1 | 88.3 |
| Pair days/litter | 50 | – | 55.6 | 76.9 | 62.5 |
| Pair days/offspring | 7 | – | 7.6 | 17.3 | 11.5 |

Source: Huber (1965).

Huber (1965) also assessed the effect of chlordecone on estrous cyclicity in virgin female mice (20/group) given either 0 or 40 ppm of chlordecone in the diet for 120 days. After 21 and 120 days of treatment, daily vaginal smears were taken for 3 to 4 weeks. In the 40 ppm females, persistent estrus appeared within 8 weeks of treatment initiation. Seventy-one percent of the smears taken in the 40 ppm females for 4 weeks after termination of chlordecone treatment were in estrus versus only 24% in controls. Huber (1965) also noted persistent estrus in 30 and 37.5 ppm female mice from group A following the reproduction test and 40 ppm female mice from group B prior to mating. The occurrence of persistent estrus is an indication that the treated female mice were under a prolonged stimulation of follicular stimulating hormone (FSH) and estrogen with insufficient luteinizing hormone stimulation. The 30 ppm treatment level represents a LOAEL for this effect.

In summary, the multiple dose reproduction study (Huber, 1965), in which male and female mice were given chlordecone in the diet for 1 month prior to mating and for 100 days following the initiation of mating, resulted in adverse reproductive effects. The 1.9 mg/kg-day dose represents a NOAEL and the 5.6 mg/kg-day dose represents a LOAEL (as determined for this review), based on a statistically significantly reduced number of mouse pairs producing a second litter.

Male and female laboratory mice (7–16 pairs per group) of mixed breeds were administered chlordecone (purity unspecified) in the diet at concentrations of 0, 10, 17.5, 25, 30, or 37.5 ppm for 1 month and then were sex paired within the same exposure grouping and placed on a normal diet throughout mating and production of offspring (Good et al., 1965). Corresponding chlordecone doses of 0, 1.9, 3.3, 4.7, 5.6, or 7.0 mg/kg-day were estimated for

males and females combined by using reference values for food consumption and body weight from U.S. EPA (1988). Reproductive indices (number of litters produced, average number of young/litter, pair days/litter, and pair days/young produced) were assessed for approximately 5 months following the initiation of mating. As shown in Table 4-9, the results suggest a dose-related effect on reproductive success (decreases in number of litters and average number of young per litter, increases in pair days per litter and per young). Though the data presented in the study were not adequate for statistical analysis (no measures of variance were provided for the reproductive parameters), visual evaluation of the data indicates a reduction in reproductive success at doses ≥ 5.6 mg/kg-day.

Table 4-9. Effects of dietary chlordecone for 1 month prior to mating on reproductive indices of male and female laboratory mice of mixed breeds

| Dietary level (ppm) | Average dose ^a (mg/kg-day) | Number of pairs | Number of litters | Number of offspring/litter | Pair days/litter | Pair days/offspring |
|---------------------|---------------------------------------|-----------------|-------------------|----------------------------|------------------|---------------------|
| 0 | 0.0 | 9 | 15 | 7.93 | 65.3 | 8.3 |
| 10 | 1.9 | 13 | 26 | 7.62 | 54.46 | 7.15 |
| 17.5 | 3.3 | 16 | 25 | 7.0 | 72.16 | 13.09 |
| 25 | 4.7 | 11 | 12 | 6.08 | 100.42 | 16.51 |
| 30 | 5.6 | 7 | 2 | 3.0 | 241.5 | 80.5 |
| 37.5 | 7.0 | 10 | 2 | 5.0 | 555.0 | 111.0 |

^aAverage doses to male and female mice (combined), based on reference values for subchronic body weight and food consumption taken from U.S. EPA (1988).

Source: Good et al. (1965).

In separate experiments by Good et al. (1965), impaired reproductive success, expressed as significantly ($p < 0.05$) reduced production of a second litter, was observed in mice that were administered chlordecone (purity unspecified) in the diet at a concentration of 5 ppm for 1 month prior to mating and for up to 5 months following initiation of mating (shown in Table 4-10). The corresponding chlordecone dose of 0.94 mg/kg-day was estimated for males and females combined by using reference values for food consumption and body weight from U.S. EPA (1988). The authors reported that continued treatment of offspring of chlordecone-treated mice with either control or 5 ppm chlordecone diets resulted in significantly reduced production of a first litter ($p \leq 0.05$), compared with untreated offspring of untreated parental mice, though reduced production of the second litter did not achieve statistical significance. The results of these studies identified a LOAEL of 0.94 mg/kg-day for impaired reproductive success; a NOAEL was not identified.

Table 4-10. Effects of dietary chlordecone (0 or 5 ppm) 1 month prior to mating and up to 5 months after initiation of mating on reproduction in BALB/c mice

| | First generation | | Second generation | | |
|------------------------------------|------------------|-----------------|-------------------|---|-----------------|
| | Control | Treated | Control | Offspring of treated mice on control diet | Treated |
| Number of pairs | 24 | 36 | 21 | 23 | 20 |
| Number of litters | 40 | 52 | 21 | 9 | 10 |
| Number of offspring | 275 | 314 | 123 | 42 | 40 |
| % producing 1 st litter | 96 | 81 | 71 | 30 ^a | 25 ^a |
| % producing 2 nd litter | 78 | 50 ^a | 29 | 9 | 15 |
| First litter size | 6.2 | 6.2 | 5.6 | 4.3 | 4.4 |
| Second litter size | 7.3 | 5.7 | 6.5 | 6.0 | 5.3 |
| Pair days/litter | 70.1 | 86 | 120 | 307 | 240 |
| Pair days/offspring | 10.2 | 14.2 | 21 | 66 | 60 |

^aReported as significant at $p < 0.05$, using binomial distribution.

Source: Good et al. (1965).

As in the previous data reported by Good et al. (1965), reproductive parameters, including litter size, pair days/litter, and pair days/young produced, were all reported as averages for the treatment or control group without any measure of variance given (i.e., standard deviation). Therefore the degree of variability for the reported reproductive parameters is unclear. Additionally, there was reduced fertility of the BALB/c untreated controls just one generation apart. For instance, 96 and 78% of untreated control animals produced first and second litters, respectively, whereas only 71 and 29% of their untreated progeny produced first and second litters. These inconsistencies limit confidence in this study and the reproducibility of the data.

In a reproductive and neurodevelopmental toxicity study, female F344 rats (10/group) were fed diets containing 0, 1, or 6 ppm of chlordecone (purity unspecified) for 60 days prior to mating (with a nonexposed male rat) through lactation day 12 (Squibb and Tilson, 1982). Corresponding maternal doses of 0, 0.1, and 0.6 mg/kg-day were estimated by using reference values for food consumption from U.S. EPA (1988) and the average of reported body weights of the dams prior to mating and on the day after parturition. Chlordecone treatment did not produce adverse effects on litter size or sex ratio of the offspring. Litters were culled to three male and three female offspring per dam on postpartum day 3. Pup body weights were similar to those of controls at 1, 7, 14, and 30 days of age, but after 100 days, body weight was significantly reduced in male pups at 0.6 mg/kg-day (19% decrease relative to controls) and female pups at 0.1 mg/kg-day (27% decrease relative to controls) and 0.6 mg/kg-day (27% decrease relative to

controls). No dose-response relationship was demonstrated in this study for decreased pup body weight. Pups were exposed to higher concentrations of chlordecone during the first 2 weeks of life (i.e., during lactation) without any significant effects on body weight. A pharmacokinetic elimination study in rats (Egle et al., 1978) demonstrated that 65.5% of an orally administered dose of chlordecone had been excreted into the feces by 12 weeks. The chlordecone body burden was assumed to be much lower at 100 days, when compared with earlier time points.

One male and one female pup from each litter were chosen at random for behavioral and pharmacological challenge testing (10 males and 10 females from each dose group) (Squibb and Tilson, 1982). The results of behavioral testing, conducted at 30 and 100 days, were primarily negative. Exposed offspring showed no statistically significant changes (compared with controls) in forelimb or hind-limb grip strength, spontaneous motor activity, startle responsiveness (air puff or acoustic stimulus), or tail-flick frequency in response to thermal stimulation. Positive results were found for one test in male offspring exposed to 6 ppm in which the animals took significantly longer time to reorient themselves to a vertical position in an assay for negative geotaxis at 100 days of age. The effect was not seen at 30 days in males and was not seen at either time point in female offspring.

The results of pharmacological challenge tests were mixed (Squibb and Tilson, 1982). Motor activity induced by subcutaneously injected 1 mg/kg apomorphine (a dopamine receptor agonist) at 114 days of age was significantly increased in male offspring of the 6 ppm group 30 minutes after dosing and male offspring of the 1 and 6 ppm groups 60 minutes after dosing. This effect was not seen in females. There was no effect on motor activity induced by d-amphetamine (a presynaptic releaser of both dopamine and norepinephrine) at 134 days in either male or female offspring. This study found little evidence of an effect of chlordecone on neurodevelopment in rats. The weight of evidence of behavioral tests was negative, except for an increased negative geotaxis latency in males at the high dose. Similarly, the positive result in the challenge test with apomorphine was observed in males, but not in females. Spontaneous motor activity of treated animals in the absence of pharmacological challenges was not different from controls. In the absence of additional effects suggesting a neurological or behavioral response, the biological significance of the alteration of dopaminergic function in chlordecone-exposed animals following pharmacological challenge is uncertain. Based on the decreased body weight of female offspring exposed gestationally and lactationally to chlordecone, a LOAEL of 0.1 mg/kg-day was determined for this review.

Adult Sherman strain male and female rats (22–25 rats/sex/group) were fed diets containing 0 or 25 ppm commercial grade chlordecone (80.6% purity) for 3 months, during which time they were housed individually and observed for clinical signs of neurotoxicity (Cannon and Kimbrough, 1979). At the end of the treatment period, selected control and chlordecone-treated male and female rats were subjected to gross and histopathologic examinations. The remaining rats (20/sex/group) were pair mated (control males with

chlordecone-treated females, control females with chlordecone-treated males, and control females with control males) during a breeding period of approximately 2 months. The production of offspring was used as an indicator of reproductive toxicity.

According to the study authors, chlordecone intake ranged from 1.62 to 1.71 mg/kg-day in 25 ppm females and from 1.17 to 1.58 mg/kg-day in 25 ppm males. Body tremors were seen in chlordecone-treated rats after 4 weeks of treatment and were most marked in treated females. At the end of the exposure period, chlordecone-treated male and female rats exhibited depressed body weight and gross and microscopic signs of adverse hepatic effects. The adrenals showed hyperplasia of the zona fasciculata and zona reticularis with marked hypertrophy of the cortex. The study authors noted gross and histopathologic signs of adverse adrenal effects in treated females. Twelve of the 20 pairs of control females and chlordecone-treated males produced offspring compared with 13/20 pairs in the controls. However, no offspring were produced among the 20 pairs of control males and chlordecone-treated females. Mating of chlordecone-treated females and control males was repeated 9 weeks after exposure cessation. Reproductive function was partially restored with 9/20 pairs producing litters, indicating some reversibility of the observed reproductive deficit in chlordecone-treated females. This study identified a LOAEL of 1.6–1.7 mg/kg-day for impaired reproductive success in female rats.

Groups of sexually mature virgin female CD-1 mice were administered chlordecone by gavage (in sesame oil) at doses of 0, 0.062, 0.125, or 0.25 mg/day (0, 2, 4, or 8 mg/kg-day), 5 days/week for 2, 4, or 6 weeks (Swartz et al., 1988). A positive control group received 17β -estradiol at a dose of 0.1 mg/day. Some mice from each group were assessed for production of oocytes (intraperitoneal administration of pregnant dam's serum gonadotropin followed 48 hours later by human chorionic gonadotropin) during the second, fourth, and sixth week of chlordecone treatment. As shown in Table 4-11, PVE was noted in most chlordecone-treated mice and positive controls as early as 2 weeks following the initiation of treatment. By week 4, all chlordecone-treated mice exhibited PVE versus 0/9 vehicle controls.

Table 4-11. Effects of chlordecone on estrous cyclicity and ovulation in CD-1 mice exposed to chlordecone by gavage 5 days/week for up to 6 weeks

| Test | Vehicle controls | Positive controls (17 β -estradiol) | Chlordecone dose (mg/kg-day) | | |
|------------------------|----------------------------------|--|------------------------------|---------------------|----------------------------------|
| | | | 2 | 4 | 8 |
| PVE ^{a,b} | | | | | |
| Week 2 | 0/9 | 7/8 | 6/8 | 5/9 | 8/9 |
| Week 3 | 0/9 | 8/8 | 6/8 | 7/9 | 9/9 |
| Week 4 | 0/9 | 7/8 | 8/8 | 9/9 | 9/9 |
| Ovulation ^c | | | | | |
| Week 2 | 19.9 \pm 2.4 (15) ^d | 30.2 \pm 11.8 (6) | 26.7 \pm 3.2 (10) | 19.2 \pm 3.2 (10) | 17.7 \pm 4.5 (15) |
| Week 4 | 28.4 \pm 2.9 (22) | 29.7 \pm 2.2 (11) | 22.9 \pm 4.3 (7) | 27.1 \pm 5.0 (6) | 14.1 \pm 2.4 (22) ^e |
| Week 6 | 23.7 \pm 2.4 (16) | 22.1 \pm 2.5 (9) | 32.4 \pm 3.8 (7) | 21.0 \pm 5.8 (7) | 14.5 \pm 3.5 (16) ^e |

^aPVE = persistent vaginal estrus, defined as the presence of epithelial cells (without leukocytes) in vaginal smears.

^bAll treatment groups for this endpoint significantly different from vehicle controls ($p < 0.05$), using the Fisher's exact test.

^cAverage number of oocytes in the oviducts at sacrifice.

^dNumber of animals.

^eStatistically significantly different from vehicle controls ($p < 0.05$) using the Student's *t*-test.

Source: Swartz et al. (1988).

After 4 and 6 weeks of treatment, ovulation in the highest chlordecone treatment group (8 mg/kg-day) resulted in statistically significantly lower numbers of ovulated oocytes relative to vehicle controls. This study identified a LOAEL of 2 mg/kg-day for PVE in virgin female CD-1 mice.

Swartz and Mall (1989) administered chlordecone (98% purity) to groups of female CD-1 mice via gavage (in sesame oil) at doses of 0 or 0.25 mg/day (8 mg/kg-day), 5 days/week for 4 weeks. A positive control group received 17 β -estradiol at a dose of 0.1 mg/day. Animals were sacrificed 24 hours following the final treatment, and the ovaries were fixed and sectioned. The abundance of small-, medium-, and large-sized follicles was determined in every tenth section. Significantly fewer small- and medium-sized follicles were found in chlordecone-treated mice relative to vehicle controls (Table 4-12). Based on observations that many of the large-sized follicles in the ovaries of chlordecone-treated mice appeared to be atretic, all histological sections of the ovaries were examined for the presence and condition of large-sized follicles. The number of large-sized follicles in chlordecone-treated mice did not differ significantly from controls; however, a significantly lower abundance of healthy large-sized follicles was noted (Table 4-12). This study identified a LOAEL of 8 mg/kg-day for adverse effects on follicle size and condition.

Table 4-12. Abundance of various-sized follicles and the condition of large-sized follicles in the ovaries of female CD-1 mice exposed to chlordecone by gavage 5 days/week for 4 weeks

| Treatment | Number of small-, medium-, and large-sized follicles ^a | | | Number of healthy and atretic large-sized follicles ^b | | |
|---------------|---|---------------------------|------------|--|-------------------------|-------------------------|
| | Small | Medium | Large | Total | Healthy | Atretic |
| Controls | 279.2 ± 39.6 | 116.2 ± 7.8 | 21.3 ± 2.5 | 58.1 ± 7.3 | 28.4 ± 6.0 | 29.7 ± 3.4 |
| 17β-Estradiol | 368.0 ± 47.5 | 231.9 ± 41.0 ^c | 28.0 ± 8.3 | 69.6 ± 6.7 | 25.4 ± 2.7 | 44.2 ± 4.3 ^c |
| Chlordecone | 190.1 ± 32.8 ^c | 103.8 ± 11.8 | 27.5 ± 3.2 | 58.7 ± 5.8 | 18.5 ± 1.9 ^c | 40.1 ± 5.1 |

^aMean ± SEM, based on evaluations of every 10th section.

^bMean ± SEM, based on evaluations of all sections.

^cStatistically significantly different from vehicle controls ($p < 0.05$) using the Student's *t*-test.

Source: Swartz and Mall (1989).

Gellert and Wilson (1979) administered chlordecone (purity unspecified; vehicle: 5% ethanol in sesame oil) to pregnant Sprague-Dawley rats by gavage at doses of 0 or 15 mg/kg-day on GDs 14–20. Untreated controls were included in the study, as well as groups of dams that were administered other pesticides. The study report did not specify the number of rats in each treatment group. The pregnant rats were allowed to deliver and raise their offspring. At 21 days of age, the offspring were sexed and weaned. At approximately 6 months of age, estrous cyclicity of female offspring was assessed via daily vaginal smears for about 2 weeks. PVE was defined as ≥ 4 consecutive days with only cornified or nucleated cells in the vaginal smear. At sacrifice immediately following assessment for estrous cyclicity, the rats were weighed and blood was collected for analysis of serum estradiol. Ovaries, uteri, and adrenals were weighed, and ovaries were histologically examined for the presence of corpora lutea. Animals with visible corpora were considered to be ovulatory. At 6 months of age, the male offspring of the treated dams were subjected to fertility testing by placing them with two experienced female rats for a period of 2 weeks. The resulting offspring of these matings were counted and sexed. At sacrifice, adrenals, testes, and ventral prostates of the F1 generation were individually weighed, and the epididymis was grossly examined for the presence of cysts.

The study authors did not report chlordecone-induced effects in the treated dams. Female offspring of the chlordecone-treated dams exhibited significantly decreased ovarian weight and significantly increased adrenal weight relative to vehicle controls, as well as significantly increased incidences of PVE (Table 4-13). In each of the control groups, all but one of the female offspring were ovulatory, whereas none of the 21 female offspring of the chlordecone-treated dams were ovulatory (Table 4-13). Serum estradiol levels in control female offspring fluctuated as expected during regular 4- or 5-day estrous cycles, whereas the levels in chlordecone-treated female offspring were observed to remain at an intermediate level. The serum estradiol levels were below 10 pg/mL in 65% of controls and 24% of the chlordecone-

treated animals. In 14% of the control animals the estradiol levels were above 47 pg/mL, whereas none of the chlordecone-treated animals had estradiol above this level. Male offspring of the chlordecone-treated dams exhibited no evidence of decreased fertility or altered sex ratios in the resulting F2 generation. This study identified a LOAEL of 15 mg/kg-day for reproductive effects in adult female offspring of rat dams administered chlordecone by gavage during GDs 14–20.

Table 4-13. Effects of chlordecone on adult female offspring of Sprague-Dawley rat dams administered chlordecone by gavage on GDs 14–20

| Treatment | Number of rats | Body weight (g) | Average weight (mg) | | | Number of rats with PVE | Number of anovulatory rats |
|-------------|----------------|----------------------|---------------------|----------|---------------------|-------------------------|----------------------------|
| | | | Ovary | Uterus | Adrenal | | |
| Control | | | | | | | |
| Untreated | 29 | 372 ± 7 ^a | 92 ± 3 | 577 ± 24 | 64 ± 1 | 2 | 1 |
| Sesame oil | 25 | 338 ± 6 | 96 ± 4 | 621 ± 26 | 68 ± 2 | 1 | 1 |
| Chlordecone | | | | | | | |
| (15 mg/kg) | 21 | 364 ± 13 | 59 ± 2 ^a | 686 ± 37 | 85 ± 3 ^a | 12 ^a | 21 ^a |

^aStatistically significantly different from sesame-oil-treated controls ($p < 0.001$).

Source: Gellert and Wilson (1979).

Several groups of investigators assessed spermatogenesis in laboratory animals that had been exposed to chlordecone. In a toxicological screen of several chemicals, chlordecone (purity unspecified) was administered to male rats of unspecified strain at dose levels of 0.625, 1.25, 2.5, 5.0, or 10.0 mg/kg/day for 10 days (U.S. EPA, 1986c). Untreated and vehicle controls were included in the study. Testes and epididymides were removed for assessment of testicular weight, sperm concentration, motility, and morphology; and histopathology. Compared with control values, alteration of sperm concentration was noted in all chlordecone-treated groups. There were no treatment-related effects on sperm motility, testosterone level, or FSH level and no testicular histopathologic findings. A LOAEL of 0.625 was identified for this study.

Linder et al. (1983) exposed male Sprague-Dawley rats (20/group) to dietary concentrations of chlordecone at 0, 5, 15, or 30 ppm for 90 days. The report does not specify how the chlordecone was added to the diet. The authors estimated the corresponding doses to be 0, 0.26, 0.83, or 1.67 mg/kg-day, respectively. After 90 days of treatment, half of the animals in each group were sacrificed for weighing and histopathological examination of the reproductive organs and measurement of epididymal sperm characteristics. Each of the remaining males in each group was bred to two untreated females over a 14-day unexposed period immediately following the 90-day exposure period. The mated females were sacrificed on GD 20, and fetal weights, fetal viability, and total number of implants were determined. The mated males were maintained for a 4.5-month recovery period prior to sacrifice and examination of sperm and

reproductive organs. Some rats in the 0.83 and 1.67 mg/kg-day groups displayed hyperexcitability and mild tremors during the treatment period. Body weight was significantly lower than that of controls by about 7% in the 1.67 mg/kg-day group at the end of treatment, but the lower dose groups were not affected. The decrease in final body weight was accompanied by significant decreases in absolute prostate and seminal vesicle weight in the 1.67 mg/kg-day group, while testis and epididymis weights were unchanged from controls. Relative weights of all of these tissues were reported to be similar to controls, although the data were not shown. No gross or microscopic pathology related to treatment was found.

Sperm viability, motility, and reserves in the right cauda epididymis were statistically significantly reduced in both the 0.83 and 1.67 mg/kg-day groups, but not at 0.26 mg/kg-day (Linder et al., 1983). The findings in the two high-dose groups were similar to each other (no increase in severity with increasing dose beyond 0.83 mg/kg-day) (see Table 4-14).

Table 4-14. Sperm parameters in male Sprague-Dawley rats following administration of chlordecone in the diet for 90 days

| Endpoint | Dose (mg/kg-day) | | | |
|---|------------------|------------|-------------------------|-------------------------|
| | 0 | 0.26 | 0.83 | 1.67 |
| Sperm motility (percent motile + SEM) | 37.0 ± 3.9 | 33.2 ± 3.8 | 19.2 ± 4.4 ^a | 22.6 ± 5.5 ^a |
| Sperm viability (percent alive + SEM) | 46.0 ± 4.7 | 36.2 ± 3.3 | 25.0 ± 3.3 ^a | 30.9 ± 4.8 ^a |
| Sperm content of right cauda epididymis (count × 10 ⁶ ± SEM) | 308 ± 14 | 290 ± 10 | 248 ± 22 ^a | 249 ± 14 ^a |

^aStatistically different from control groups according to ANOVA ($p < 0.05$).

Source: Linder et al. (1983).

Neither sperm morphology nor sperm count in the epididymal fluid was affected at any dose. Reproductive performance (determined by number of pregnant females, number of live litters, average live litter size, number of implants, percentage of resorptions, and fetal weight) was similar in exposed and control groups. No effects of any type were found after the 4.5-month recovery on control diet. In this study, subchronic dietary exposure to ≥ 0.83 mg/kg-day produced significant reductions in sperm motility, viability, and reserves without affecting sperm morphology and sperm count in the epididymal fluid or without affecting male reproductive performance. Similar effects (oligospermia in the absence of a reduction in reproductive performance) have also been observed in occupationally exposed humans (Guzelian, 1982a, b; Guzelian et al., 1980; Taylor et al., 1978). Doses of ≥ 0.83 mg/kg-day also produced neurological effects (hyperexcitability and tremors) in the rats, while these effects were not observed in the 0.26 mg/kg-day dose group. This study identified a LOAEL of 0.83 mg/kg-day and a NOAEL of 0.26 mg/kg-day, based on the occurrence of neurological effects and statistically significant spermotoxic effects.

Additional reproductive studies exist that evaluate the effect of acute injected chlordecone (at doses of 20–80 mg/kg) in experimental animals. Effects observed in these studies were similar to studies of repeat oral administration of chlordecone and generally included changes in estrous cyclicity and fertility (Williams and Uphouse, 1991; Johnson et al., 1990; Pinkston and Uphouse, 1987–1988). These acute injection studies provide information to support reproductive effects at high doses of chlordecone but do not generally contribute additional dose-response information regarding the most sensitive effects of chlordecone exposure.

4.3.2. Developmental Toxicity Studies

The developmental toxicity of chlordecone in humans is not known. Chlordecone produces developmental toxicity in rats and mice at dose levels that also produce maternal toxicity (Seidenberg et al., 1986; Chernoff and Rogers, 1976). Chernoff and Rogers (1976) administered chlordecone (purity unspecified) to groups of pregnant CD rats at gavage doses of 0, 2, 6, or 10 mg/kg-day on GDs 7–16. Dams were observed for clinical signs and weight gain, and sacrificed on GD 21 for assessment of liver/body weight and evaluation of fetuses. Fetal parameters evaluated include number of implants, mortality, weight, and gross developmental abnormalities. Study results are depicted in Table 4-15. Significant maternal toxicity was observed in high-dose dams. All groups of dosed dams exhibited significantly depressed weight gain, and the average liver/body weight ratio was significantly increased in the two highest dose groups (6 and 10 mg/kg-day). Fetotoxicity was observed as significantly depressed fetal body weight and delayed ossification in 6 and 10 mg/kg-day dose groups and significantly increased incidences of litters with fetuses having enlarged renal pelvis, edema, undescended testes, or enlarged cerebral ventricles in the 10 mg/kg-day group relative to controls. The study identified a LOAEL of 2 mg/kg-day for maternal toxicity, based on significantly depressed maternal body weight gain (16% lower than controls). The study identified a NOAEL of 2 mg/kg-day and a LOAEL of 6 mg/kg-day for fetotoxicity.

Table 4-15. Maternal and fetal effects following gavage dosing of pregnant rat dams with chlordecone on GDs 7–16

| | Dose level (mg/kg-day) | | | |
|---|------------------------|-------------------------|-------------------------|-------------------------|
| | 0 | 2 | 6 | 10 |
| Maternal effects ^a | | | | |
| Number inseminated | 26 | 31 | 35 | 42 |
| Maternal deaths | 0 | 1 | 0 | 8 ^b |
| Number pregnant at sacrifice | 23 | 24 | 33 | 30 |
| Weight gain (g) | 73.5 ± 3.7 | 62.4 ± 2.9 ^b | 33.8 ± 2.4 ^b | 34.0 ± 5.6 ^b |
| Liver/body weight | 5.0 ± 0.1 | 5.1 ± 0.1 | 5.9 ± 0.1 ^b | 7.4 ± 0.2 ^b |
| Fetal effects ^a | | | | |
| Implants/dam | 10.2 ± 0.4 | 10.4 ± 0.6 | 11.0 ± 0.4 | 9.1 ± 0.5 |
| Percent mortality | 9.5 ± 3.0 | 8.1 ± 2.7 | 6.5 ± 1.2 | 17.7 ± 4.9 |
| Weight at sacrifice (g) | 4.1 ± 0.1 | 4.0 ± 0.1 | 3.9 ± 0.1 ^b | 3.7 ± 0.1 ^b |
| Sternal ossification centers | 5.6 ± 0.1 | 5.5 ± 0.1 | 5.3 ± 0.1 | 5.3 ± 0.1 |
| Caudal ossification centers | 4.7 ± 0.1 | 4.5 ± 0.1 | 4.4 ± 0.1 ^b | 4.0 ± 0.2 ^b |
| Percent supernumerary ribs | 24.4 ± 6.3 | 28.1 ± 5.5 | 24.5 ± 5.4 | 17.4 ± 3.9 |
| Enlarged renal pelvis | 1 | 2 | 5 | 10 ^b |
| Edemac | 0 | 1 | 0 | 10 ^b |
| Undescended testis | 0 | 0 | 1 | 5 ^b |
| Enlarged cerebral ventricles ^c | 0 | 0 | 0 | 5 ^b |

^aMean ± SE.

^bStatistically significantly different from controls ($p < 0.05$).

^cNumber of litters with one or more fetuses exhibiting the effect.

Source: Chernoff and Rogers (1976).

Chernoff and Rogers (1976) also administered chlordecone (purity unspecified) to groups of pregnant CD-1 mice at gavage doses of 0, 2, 4, 8, or 12 mg/kg-day on GDs 7–16. Dams were observed for clinical signs and weight gain, and sacrificed on GD 18 for assessment of liver and body weight and evaluation of fetuses. Maternal and fetotoxicity were assessed in the same manner as that described for the rats. In mice, significantly depressed maternal weight gain was noted at 8 and 12 mg/kg-day, and all dose groups exhibited significantly increased maternal liver and body weight (Table 4-16). Signs of fetotoxicity were observed only in the highest dose group and consisted of significantly increased fetal mortality. The study identified a LOAEL of 2 mg/kg-day for maternal toxicity, based on a statistically significant 10% increase in relative liver weight in the 2, 4, and 8 mg/kg-day dose groups. The study identified a NOAEL of 8 mg/kg-day and a LOAEL of 12 mg/kg-day for fetotoxicity. The fetal effects may have been the direct result of maternal toxicity since they occurred at doses that were toxic to the dams.

Table 4-16. Maternal and fetal effects following gavage dosing with chlordecone on GDs 7–16

| | Dose level (mg/kg-day) | | | | |
|-------------------------------|------------------------|------------------------|------------------------|------------------------|--------------------------|
| | 0 | 2 | 4 | 8 | 12 |
| Maternal effects ^a | | | | | |
| Number inseminated | 26 | 16 | 24 | 25 | 12 |
| Maternal deaths | 0 | 0 | 0 | 0 | 1 |
| Number pregnant at sacrifice | 16 | 14 | 16 | 19 | 5 |
| Weight gain (g) | 4.3 ± 0.5 | 4.1 ± 0.4 | 3.3 ± 0.4 | 0.7 ± 0.9 ^b | -2.8 ± 0.9 ^b |
| Liver/body weight | 6.8 ± 0.3 | 7.5 ± 0.2 ^b | 7.9 ± 0.1 ^b | 8.6 ± 0.3 ^b | 7.6 ± 0.6 |
| Fetal effects ^a | | | | | |
| Implants/dam | 12.8 ± 0.6 | 12.0 ± 0.8 | 12.4 ± 0.7 | 11.3 ± 0.7 | 11.8 ± 1.4 |
| Percent mortality | 15.6 ± 3.3 | 12.4 ± 3.5 | 11.8 ± 2.1 | 16.9 ± 5.1 | 53.4 ± 19.4 ^b |
| Weight at sacrifice (g) | 1.0 ± 0.1 | 1.0 ± 0.1 | 1.1 ± 0.1 | 1.0 ± 0.1 | 1.3 ± 0.1 |
| Sternal ossification centers | 5.5 ± 0.1 | 5.3 ± 0.2 | 5.6 ± 0.2 | 5.1 ± 0.3 | 6.0 ± 0.0 |
| Caudal ossification centers | 4.0 ± 0.3 | 3.5 ± 0.5 | 4.5 ± 0.4 | 4.1 ± 0.5 | 6.4 ± 0.4 |
| Percent supernumerary ribs | 33.0 ± 6.8 | 20.9 ± 9.4 | 13.8 ± 5.1 | 26.2 ± 6.4 | 12.3 ± 4.8 |

^aMean ± SE.

^bStatistically significantly different from controls ($p < 0.05$).

Source: Chernoff and Rogers (1976).

Additional developmental studies exist on chlordecone administered by injection (of 5–100 mg/kg) during gestation or postnatally. Effects associated with chlordecone exposure generally included alterations in neurological function, as well as impaired learning and behavioral changes, alterations in sexual differentiation, and weak estrogenic effects (Laessig et al., 2007 Sierra and Uphouse, 1986; Cooper et al., 1985; Mactutus and Tilson, 1985; Rosecrans et al., 1985). These acute injection studies help provide information to support developmental effects at high doses of chlordecone, but do not generally contribute additional dose-response information regarding the most sensitive effects of chlordecone exposure during development.

4.3.3. Screening Studies

In a neonatal survival screen, chlordecone (purity unspecified) was administered to pregnant F344 rats at a gavage dose level of 0 or 10.0 mg/kg-day during GDs 7–16 (U.S. EPA, 1986c). Neonatal survival was assessed on days 1 and 3 postpartum. Significantly ($p < 0.5$) reduced survival was noted on day 3 (but not day 1) postpartum (U.S. EPA, 1986c). In a

developmental toxicity screen in the ICR/SIM mouse, chlordecone was administered by gavage at a dose of 0 or 24 mg/kg-day during GDs 8–12 (Seidenberg et al., 1986). Maternal toxicity was observed with decreased body weight gain and mortality in 18% of treated dams. Decreases were also observed in neonatal body weight gain and percent survival (Seidenberg et al., 1986).

4.4. OTHER DURATION-OR ENDPOINT-SPECIFIC STUDIES

4.4.1. Acute Toxicity Studies

Oral LD₅₀ values for chlordecone range from 71 mg/kg-bw for rabbits to 250 mg/kg-bw for dogs (Larson et al., 1979a). The oral LD₅₀ value for rats is 125 mg/kg-bw (Gaines, 1969). In experimental animals, the systemic effects of chlordecone following short-term exposures generally include nervous system effects (tremor and hyperexcitability), reproductive system toxicity (effects on estrous cyclicity and sperm parameters), liver effects (hypertrophy, microsomal enzyme induction, and ultrastructural changes), musculoskeletal effects (resulting from alterations in ATPase activity and calcium homeostasis), and thyroid and adrenal effects (ATSDR, 1995; U.S. EPA, 1986c; WHO, 1984). The effects observed in the liver following exposure to chlordecone are similar to those generally produced by halogenated hydrocarbons; these effects include increase in liver weight or size and induction of the mixed function oxidase enzyme system (ATSDR, 1995). Chlordecone was also shown to alter lipid storage and metabolism in mice (Carpenter et al., 1996; Chetty et al., 1993a, b), and hepatobiliary excretion of certain chemicals was inhibited by chlordecone following acute exposure (see Section 3.4).

Other systemic effects reported following acute chlordecone exposure include decreases in food intake and body weight gain (ATSDR, 1995; Williams et al., 1992; U.S. EPA, 1986c; Chernoff and Kavlock, 1982; Chernoff and Rogers, 1976), altered thermoregulation resulting in a decrease in core temperature that persisted for up to 12 days following ingestion of a single dose of 55 or 75 mg/kg in rats (Swanson and Woolley, 1982), and slight hyperthermia in rats following 12 weeks of exposure at 7.1 mg/kg-day (Pryor et al., 1983). The cardiovascular effects in rats after acute-duration exposure to chlordecone are limited to biochemical changes in cardiac tissue, such as membrane enzyme inhibitions and altered protein phosphorylation (Kodavanti et al., 1990); however, the toxicological implications of these changes are unknown.

4.4.2. Potentiation of Halomethane Toxicity

Laboratory studies of chlordecone potentiation of halomethane liver toxicity provide insight into potential mechanisms of chlordecone-induced liver toxicity, though doses used in these studies are not considered environmentally relevant.

Chlordecone potentiates the liver toxicity and lethality of carbon tetrachloride (CCl₄) and other halomethanes (e.g., chloroform, bromotrichloromethane) in rats and mice, and this interaction has been widely studied and reviewed (Mehendale, 1994, 1990; Mehendale et al., 1989; Plaa et al., 1987; Curtis et al., 1981). The exposure of rats to 10 ppm chlordecone in the

diet for 15 days greatly increased the liver toxicity of halomethanes, leading to hepatic failure and death (Soni and Mehendale, 1993). Liver toxicity was generally demonstrated by measurement of elevated serum enzyme activities and histopathological changes, including necrosis, lipid accumulation, and hepatocyte swelling. This effect was specific to chlordecone and was not observed following pretreatment with other organochlorine pesticides (e.g., mirex and photomirex).

Chlordecone enhanced the oxidative metabolism of halomethanes; however, enzyme induction was not correlated with the potentiation of liver toxicity. More efficient enzyme inducers, such as phenobarbital, did not significantly potentiate the toxicity of CCl₄ (Mehendale and Klingensmith, 1988; Curtis et al., 1981). Chlordecone appears to enhance the liver toxicity of halomethanes by suppressing the hepatocellular regeneration that is required to repair liver injury and restore hepatolobular architecture and function (Kodavanti et al., 1992; Mehendale, 1989, 1990). Partially hepatectomized rats are protected from chlordecone-CCl₄ toxicity because of an increase in the rate of cell turnover as measured by [³H]-thymidine incorporation into hepatocellular DNA and an increase in the percentage of mitotic figures (Kodavanti et al., 1989; Young and Mehendale, 1989). Protection from liver toxicity was also provided by pretreatment with cyanidanol, which stimulated hepatocellular regeneration as evidenced by increased [³H]-thymidine incorporation (Soni and Mehendale, 1991a, b, c). Polyamine metabolism was inhibited by cotreatment with chlordecone and bromotrichloromethane (Rao et al., 1990). Polyamines are important for the cell growth and proliferation process that results in liver regeneration and repair.

The chlordecone suppression of liver cell regeneration and repair may be related to the compromised energy status of hepatocytes in animals exposed to chlordecone. Treatment of rats with chlordecone and CCl₄ caused a decrease in liver ATP levels and an inhibition of oligomycin-sensitive Mg²⁺-ATPase (Kodavanti et al., 1990). Chlordecone affects calcium homeostasis in hepatocytes, leading to a decline in glycogen storage and a reduced energy status (Kodavanti et al., 1993, 1990). Chlordecone-CCl₄ administration caused an inhibition in microsomal and mitochondrial calcium uptake and a decrease in the high affinity component of hepatic plasma membrane Ca²⁺-ATPase. Administration of fructose 1,6-diphosphate to rats resulted in protection from chlordecone-CCl₄ hepatotoxicity due to an increase in the levels of liver cell ATP (Rao and Mehendale, 1989). ATP administration during the early phase of liver injury also helped to restore normal liver function through enhanced regeneration and repair (Soni and Mehendale, 1991a, b, c).

Several studies have indicated an age-related susceptibility to the chlordecone potentiation of CCl₄ hepatotoxicity (Dalu et al., 1995; Cai and Mehendale, 1993). Developing rats have been shown to be resistant to the lethal effects of the chlordecone-CCl₄ combination treatment. Postnatal rats recovered more quickly from CCl₄-induced liver injury than young adult rats, due to the higher level of ongoing cell division and an additional stimulatory response

to liver injury (measured by [³H]-thymidine incorporation into hepatocellular DNA). The resiliency of postnatal rats was abolished by administration of the antimetabolic agent colchicine, highlighting the importance of cell turnover in liver tissue repair (Dalu et al., 1998). Aged rats (2 years old) were also shown to be resistant to the potentiation of CCl₄ liver toxicity by chlordecone due to the robust and early liver tissue repair in old rats as compared with young adult rats (3 months) (Murali et al., 2002). Gender effects were noted, with female rats being more sensitive to chlordecone-CCl₄ hepatotoxicity than male rats (Blain et al., 1999).

4.4.3. Neurotoxicity Studies

With tremor being a cardinal feature of chlordecone intoxication in humans, research into the mode of action of the neurological changes has been the focus of several studies. A number of studies have associated alterations in neurotransmitter activity (e.g., α -noradrenergic, dopaminergic, and serotonergic systems) with chlordecone-induced tremor and exaggerated startle response (Vaccari and Saba, 1995; Brown et al., 1991; Herr et al., 1987; Desaiyah, 1985; Hong et al., 1984; Fujimori et al., 1982b). At the cellular level, changes in ATPase activity and calcium homeostasis in the nervous system have been related to chlordecone exposure across species (ATSDR, 1995). The reported effects of chlordecone exposure on calcium balance in whole animal studies include decreased calcium uptake in rats following a single oral dose of 40 mg/kg (End et al., 1981); decreased total protein-bound, myelin, and synaptosomal calcium following eight consecutive daily oral doses of 25 mg/kg-day in 4- to 6-week-old male ICR mice (Hoskins and Ho, 1982); decreased total protein-bound and mitochondrial calcium content with increased nuclear calcium content in 24-week-old male ICR mice following a single oral dose of 25 mg/kg (Hoskins and Ho, 1982); and decreased brain calmodulin in rats exposed to 2.5 mg/kg-day orally for 10 consecutive days (Desaiyah et al., 1985). In vitro studies have supported that chlordecone may alter calcium regulation of neuronal function (Bondy and McKee, 1990; Vig et al., 1989; End et al., 1981).

4.4.4. Endocrine Disruption Studies

Specific mechanisms of chlordecone-induced reproductive effects are not known, although it is believed that an estrogenic mode of action may be involved. Observed chlordecone-induced reproductive effects include oligospermia, reduced sperm motility, and decreased libido in occupationally exposed males (Taylor, 1985, 1982; Guzelian, 1982a; Taylor et al., 1978) and decreased offspring production in laboratory animals (Cannon and Kimbrough, 1979; Good et al., 1965; Huber, 1965). Testicular atrophy, altered sperm characteristics, persistent vaginal estrus, and anovulation observed in chlordecone-treated laboratory animals mimic similar effects produced by excessive estrogen (Swartz et al., 1988; U.S. EPA, 1986c; Uphouse, 1985; Linder et al., 1983; Larson et al., 1979a; Huber, 1965). Estrogens can alter gene expression in reproductive tissues through interaction with nuclear estrogen receptors.

Mechanistic studies, therefore, have been designed to assess the potential of chlordecone to mimic the action of estrogen.

In cell-free preparations containing rat uterine estrogen receptors, 8 μM chlordecone inhibited the binding of [^3H]estradiol by nearly 50% (Bulger et al., 1979). It was further demonstrated that chlordecone caused the translocation of estrogen receptors from the cytosolic to the nuclear fraction in both isolated rat uteri and ovariectomized immature rats. These results indicate that chlordecone may act directly on the uterus. In another study, chlordecone-induced uterine effects observed in ovariectomized immature rats were enhanced by coadministration of estradiol, an indication that chlordecone and estradiol act at the same site in uterine tissue (Johnson, 1996). Chlordecone demonstrated a relatively high affinity for recombinant human estrogen receptors; 5.7 μM (Bolger et al., 1998) and 9 μM chlordecone (Scippo et al., 2004) caused 50% inhibition of 17β -estradiol binding. Chlordecone exhibits approximately equal affinity for both subtypes of human estrogen receptors ($\text{ER}\alpha$ and $\text{ER}\beta$) (Kuiper et al., 1998). In one study, uterine levels of adenosine 3'5'-cyclic monophosphate (cAMP) decreased with increasing uterine weight following repeated exposure to chlordecone in ovariectomized immature rats (Johnson et al., 1995). The levels of cAMP were not decreased in similarly treated rats that were also given an antiestrogen (ICI-182,780), indicating that the chlordecone-induced effect on cAMP is estrogen receptor-dependent.

The affinity of chlordecone for estrogen appears to be tissue-dependent. Although competition between [^3H]estradiol and chlordecone was comparable in magnitude within estrogen receptor preparations from brain or uterine tissues of rats, *in vivo* binding of chlordecone in the brain of ovariectomized rats was much less than that observed in the uterus (Williams et al., 1989). The basis for this *in vivo* tissue-specific difference is not clear but may result, at least in part, from a greater time requirement for chlordecone to reach a concentration in the brain that could result in a significant estrogenic effect. Furthermore, although chlordecone may mimic the effect of estrogen in uterine tissue, chlordecone appears to function as an estrogen antagonist in central nervous tissue (Huang and Nelson, 1986; Uphouse et al., 1986).

Chlordecone interacts *in vitro* and *in vivo* with the estrogen receptor system in the rat uterus. Hammond et al. (1979) found that it competes with estradiol for binding to the cytoplasmic receptor *in vitro* and also induces nuclear accumulation of estrogen receptor sites in uteri *in vitro*. Chlordecone translocates estrogen receptor sites to the uterine nucleus, increases uterine weight, and stimulates the synthesis of the progesterone receptor when it is injected into immature female rats (Hammond et al., 1979). Chlordecone has been shown to increase growth of rat leioma cell leiomyoma, however, not to the extent of estradiol (Hodges et al., 2000).

Results of a recent study indicate that chlordecone-induced uterine effects may also be induced via a pathway other than that which includes the estrogen receptor. Chlordecone up-regulated uterine expression of an estrogen-responsive gene, lactoferrin, in $\text{ER}\alpha$ knockout mice,

whereas these effects were not elicited by 17 β -estradiol (Das et al., 1997). Neither the estrogen receptor antagonist ICI-182,780 nor 17 β -estradiol inhibited the chlordecone-induced uterine expression of lactoferrin in these mice.

Chlordecone has been tested for its potential to bind to other receptors. The chemical exhibited relatively high affinity for recombinant human progesterone receptors (Scippo et al., 2004); 11 μ M chlordecone resulted in 50% inhibition of progesterone binding. Treatment with chlordecone in ovariectomized (NBZ \times NZW) F_1 mice have indicated diminished prolactin levels in contrast with estrogen treatment which elevates prolactin levels (Wang et al., 2007). Chlordecone exhibited characteristics of a partial androgen antagonist, based on 50% reduction of inhibition of 5 α -dihydroxytestosterone-mediated activation of luciferase activity by 6.9 μ M chlordecone in the human PC-3 prostate carcinoma cell line (Schrader and Cooke, 2000).

4.4.5. Immunological Studies

Several studies have examined the potential for general immunotoxicity associated with chlordecone exposure, and two studies have investigated chlordecone effects on the acceleration of an autoimmune disease. Smialowicz et al. (1985) exposed male F344 rats to technical grade chlordecone (87% pure) in corn oil by gavage for 10 days at doses of 0.625, 1.25, 2.5, 5.0, and 10 mg/kg-day (10 rats/dose). Dose groups also included a vehicle control group (corn oil), an untreated cage-matched control group, and cyclophosphamide (1.5–24 mg/kg-day) exposure groups as positive controls for immunosuppression. Blood samples were taken for total and differential white blood cell counts, and the spleen and thymus weights were recorded. Single cell suspensions were prepared from the spleen, and the lymphoproliferative response of splenocytes to the T-cell mitogens phytohemagglutinin (PHA) and concanavalin A (con A), the T- and B-cell mitogen pokeweed mitogen, and the B-cell mitogen *Salmonella typhimurium* mitogen (STM) were assayed. A single functional immune test assessing natural killer (NK) cell activity of splenocytes was also performed. NK activity was measured against W/Fu-G1 rat lymphoma cells and YAC-1 mouse lymphoma cells as the target cell population. The high dose (10 mg/kg-day) of chlordecone caused a 20% reduction in body weight as well as reduced relative spleen and thymus weights (8 and 24% respectively). The high dose was also associated with a 69% reduction in the concentration of circulating neutrophils, but no change was seen in the number of lymphocytes, monocytes, or overall leukocytes. A reduced mitogenic response to PHA was observed only in the 2.5 mg/kg-day chlordecone group. The high dose of chlordecone was associated with a 45% reduced mitogenic response to con A, a 66% increased mitogenic response to STM, and an almost threefold increase in background mitogenic response. In rats exposed to the high dose of chlordecone, NK cell activity was reduced by 62–73% against both target cell lines. The authors suggested that the observed effects in the high-dose animals (10 mg/kg-day) were due to overt toxicity. The authors also noted that at 10 mg/kg-day, rats displayed tremors characteristic of chlordecone intoxication, and therefore the decreased body

weight, decreased spleen and thymus weight, altered lymphoproliferative response, and decreased NK cell activity may have been secondary to overt toxicity.

The effects of chlordecone exposure on antibody response were examined as part of a study of the consequences of malnutrition on antibody response in male Sprague-Dawley rats (Chetty et al., 1993c). For the purpose of this review, only data from the control and chlordecone-treated rats fed nutritionally sufficient diets are presented. Rats (six/group) were exposed to 0, 10, or 100 ppm (doses calculated as 0, 0.96, or 9.6 mg/kg-day)³ chlordecone in the diet for 2 or 4 weeks. Rats were immunized by injection of sheep red blood cells (SRBCs) 4 days before the end of chlordecone exposure. In addition to measuring body weight, the authors measured spleen weight and antibody response to SRBCs as determined by the plaque-forming cell (PFC) assay. Chlordecone exposure for either 2 or 4 weeks increased the PFC response. Although the results are only presented graphically, dietary exposure of 10 ppm chlordecone increased the PFC response about two- to threefold over controls. At this dose, chlordecone treatment significantly reduced body weight by 15% and increased relative spleen weight by 29%. Average body weight and spleen weight were not reported for animals exposed to 100 ppm.

No additional studies of general immunotoxicity of chlordecone were found. As part of an acute neurotoxicity study, however, a single dose of 75 mg/kg chlordecone to Sprague-Dawley rats resulted in significant reductions in thymus weights (Swanson and Woolley, 1982). As with the results from Smialowicz et al. (1985), the dose associated with thymus weight reduction was also associated with overt toxicity.

Several studies from the same laboratory have investigated the potential effects of chlordecone treatment on autoimmune disease (Wang et al., 2007; Sobel et al., 2006, 2005). Sobel et al. (2005) investigated the effect of chlordecone in female (NZB × NZW)_{F1} mice, a murine model of systemic lupus erythematosus in which the principal clinical manifestation of lupus is renal disease, specifically immune-mediated glomerulonephritis. In this study, female 8-week-old (NZB × NZW)_{F1} control, ovariectomized, or sham-operated mice were implanted with 60-day sustained-release pellets containing doses of 0, 0.01, 0.1, 0.5, or 1 mg chlordecone (99.2% pure). Pellets were replaced every 60 days throughout the experiment. For this phase of the experiment, treatment groups consisted of 10 animals/group, whereas the control group consisted of 20 animals. Urine protein, blood urea nitrogen (BUN), and body weight were evaluated monthly for all animals. Mice were euthanized at the conclusion of the experiment if BUN exceeded 50 mg/dL or if proteinuria exceeded 2,000 mg/dL. IgG double-strand DNA antibody (anti-dsDNA) titers in serum of some treatment groups were determined by indirect enzyme-linked immunosorbent assay (ELISA). Kidneys were removed for histological examination and glomerular damage was scored by light microscopy. Additionally, a subset of

³Calculation: mg/kg-day = (ppm in feed × kg food/day)/kg-bw. Reference food consumption rates of 0.0179 kg/day (U.S. EPA, 1988) and reported average body weight of 0.188 kg (males) were used.

treatment groups were examined for IgG-mediated immune complex deposition in glomeruli by using immunohistofluorescence.

Mice treated with 1.0 or 0.5 mg chlordecone pellets developed renal disease significantly earlier than did ovariectomized controls ($p < 0.05$). This observation was also correlated with proteinuria and the early appearance of immune complex glomerulonephritis. Additionally, mice treated with chlordecone developed elevated anti-dsDNA titers earlier than ovariectomized controls. Immunohistofluorescence analysis of renal sections from a subset of animals treated for 8 weeks with 1 mg chlordecone showed enhanced deposits of IgG immune complexes as compared with untreated controls. The lowest dose per pellet found to produce a significant decrease in time to onset of renal disease was found to be 0.5 mg. Based on average body weight, the authors calculated a dosing rate per unit body weight of 0.2 mg/kg-day. However, blood levels of chlordecone were not examined, and the equivalent oral dose needed to achieve this effect is uncertain.

After the demonstration that chronic chlordecone exposure accelerates the development of autoimmunity in ovariectomized female (NZB \times NZW) F_1 mice (Sobel et al., 2005), additional studies were designed to examine the effect of chlordecone on autoimmunity and renal disease in ovary-intact female (NZB \times NZW) F_1 mice and female BALB/c mice, a mouse strain that is not predisposed to the development of autoimmune-related renal disease (Sobel et al., 2006). As in the previous study, 8-week-old female mice were implanted with 60-day sustained-release pellets containing 0, 0.001, 0.01, 0.1, 0.5, 1, or 5 mg chlordecone subcutaneously above the shoulders. Blood and urine were collected once per month for the assessment of renal function by BUN analysis and urine protein content. Mice were euthanized at the conclusion of the experiment if BUN exceeded 50 mg/dL or if proteinuria exceeded 2,000 mg/dL. Blood was taken for serum analysis and kidneys were removed for later histological analysis by light microscopy. Antigen-specific antibody levels for anti-dsDNA and antichromatin were determined by indirect ELISA.

In the first half of the experiment, involving chlordecone treatment in ovary-intact (NZB \times NZW) F_1 mice, Sobel et al. (2006) reported that chlordecone shortened survival, decreased the time to onset of elevated autoantibody titers, and accelerated glomerulonephritis in a dose-dependent manner. Median survival of control groups was 25 weeks, compared with 21 and 18 weeks in mice implanted with the 1 mg and 5 mg chlordecone pellets, respectively. Survival curves for mice treated with chlordecone were significantly different from controls by log rank test for trend ($p = 0.01$). Time to development of renal disease in mice treated with the 5 mg pellets was significantly shorter than in controls ($p < 0.05$). However, histopathology associated with renal disease was similar between the treated and untreated groups. Mice implanted with either 1 or 5 mg chlordecone pellets developed anti-dsDNA and antichromatin autoantibody titers significantly earlier than controls ($p \leq 0.005$).

In the second half of the experiment, involving chlordecone treatment of BALB/c mice, Sobel et al. (2006) performed the same assays as for the (NZB \times NZW) F_1 mice. No treatment-

related effects were seen in mortality, and none of the chlordecone-exposed BALB/c mice developed renal disease. Autoantibody titers (anti-dsDNA and antichromatin) were not different from controls. Total serum IgG2a and IgG1 were statistically increased in mice implanted with the 1 and 5 mg chlordecone pellets ($p < 0.01$). The failure of chlordecone to induce renal disease or autoantibodies in BALB/c mice (a strain not predisposed to the development of autoimmunity or renal disease) emphasizes the importance of genetic background on the effects of chlordecone on autoimmunity.

The mechanism by which chlordecone accelerates autoimmunity in female (NZB \times NZW) F_1 mice is unknown. The (NZB \times NZW) F_1 mouse is a model of systemic lupus erythematosus, an autoimmune disorder that affects women more frequently than men (Lahita, 1997). Estrogen receptor binding may play a role in some forms of autoimmune disease in rodents and humans (Ahmed et al., 1999), and, in the (NZB \times NZW) F_1 mouse model of systemic lupus erythematosus, 17 β -estradiol accelerates the development of glomerulonephritis with similar results to the effects observed following chlordecone treatment (Sobel et al., 2005). Sobel et al. (2005) hypothesized that chlordecone's acceleration of autoimmunity may be related to its estrogenic properties and ability of chlordecone to bind the estrogen receptor. However, the poor correlation between autoimmune effects and estrogenic activity of chlordecone as measured by uterine hypertrophy suggests that a non-estrogen-receptor-mediated mechanism may be important (Sobel et al., 2005). Further studies by this lab have supported mechanisms of autoimmune effects in this model system which are distinct from estradiol (Wang et al., 2008, 2007). An additional study by the same laboratory was performed to compare the mechanism of chlordecone-accelerated autoimmunity to that of 17 β estradiol-accelerated autoimmunity in (NZB \times NZW) F_1 mice by examining gene and protein expression of B cells (Wang et al., 2007). As with the earlier experiments, 6–8-week-old ovariectomized female (NZB \times NZW) F_1 mice were implanted with 60-day sustained-release pellets. In this experiment, pellets contained 1 mg chlordecone, 5 mg chlordecone, 0.05 mg estradiol, or matrix only for controls. Mice were euthanized 5–6 weeks after implantation in order to evaluate the development of autoimmune pathology rather than overt effects. Spleens were removed and splenic tissue and cells were prepared for analysis. Splenocytes were analyzed for proliferation, apoptosis, and mRNA and cDNA expression. The following immunological markers were analyzed for expression: B220, IgM, CD19, CD21, CD24, CD44, CD69, CXCR4, CXCR5, ICAM-1, VCAM-1, MHC II, B7.2, and GL7. The authors stated that germinal center activity (the area in the lymph nodes where B lymphocytes rapidly divide) and cell surface markers of B cells were examined because of the importance of the germinal center in negative selection for autoreactive B cells. Both chlordecone exposure and estradiol treatment activated splenic B cells and enhanced germinal center activity as shown by upregulated protein expression of GL7, CXCR5, and CXCR4. Both treatments also resulted in reduced B cell apoptosis and increased patterns of protein and gene expression that may increase survival of autoreactive B cells (i.e., B cell expression of ICAM-1

and VCAM-1 cell adhesion molecules and Bcl-2 and shp-1 gene expression in B cells from the germinal centers). However, major differences were also observed between the effects of chlordecone exposure and that of estradiol, particularly in the lack of an effect of chlordecone on splenic B cell subsets such as CD138⁺B220⁻ populations. The authors concluded that differences in the effects between chlordecone and estradiol indicate that chlordecone does not accelerate the development to systemic lupus erythematosus by functioning strictly as an estrogen mimic.

4.5. MECHANISTIC DATA AND OTHER STUDIES IN SUPPORT OF THE MODE OF ACTION

4.5.1. Genotoxicity

The weight of evidence from in vivo and in vitro studies suggests that chlordecone is not mutagenic. The majority of studies have not shown genotoxic activity in a variety of short-term in vitro assays. There is no evidence that chlordecone is a mutagen in *S. typhimurium* or *Escherichia coli* (Mortelmans et al., 1986; U.S. EPA, 1986c; Probst et al., 1981; Schoeny et al., 1979). Further, chlordecone alcohol, the major metabolite of chlordecone in humans, is not mutagenic in *S. typhimurium* (Mortelmans et al., 1986). Chlordecone also gave negative results when tested for enhancement of unscheduled DNA synthesis in primary cultures of adult rat hepatocytes (Probst et al., 1981; Williams, 1980). The clastogenic activity of chlordecone is unclear. Chlordecone was investigated for potential clastogenic activity in Chinese hamster ovary (CHO) cells (Galloway et al., 1987; Bale, 1983). Bale (1983) reported that chlordecone treatment of CHO (M3-1) cells (2, 4, or 6 µg/mL) produced chromosome breaks, chromatid breaks, dicentric chromosomes, and chromosome interchanges. In a later study employing higher doses, chlordecone did not increase the frequency of CHO cells with abnormal chromosome morphology over a nonactivated concentration range of 10–20 µg/L or over an Aroclor 1254-induced rat liver S9-activated concentration range of 5–15 µg/L (Galloway et al., 1987).

There has been limited testing of chlordecone in whole-animal genotoxicity assays. The available data generally show that chlordecone is not mutagenic in whole-animal tests. Chlordecone was not clastogenic in male Sprague-Dawley rat germinal cells in a dominant lethal assay at doses of 3.6 or 11.4 mg/kg-day orally for 5 consecutive days (Simon et al., 1986, 1978). Although chlordecone increased ornithine decarboxylase activity (indicative of cellular proliferation) in rat livers following oral exposure, it did not induce DNA damage in the target organ (Mitra et al., 1990; Kitchin and Brown, 1989).

4.5.2. Tumor Promotion and Mechanistic Studies

Chlordecone was tested in a two-stage model of liver carcinogenesis in both male and female Sprague-Dawley rats (Sirica et al., 1989). Male rats were subjected to two-thirds hepatectomy and 24 hours later were administered a single gavage dose (20 mg/kg) of the

initiator chemical diethylnitrosamine (DEN) in water. Ten days following initiation, rats began to receive biweekly s.c. injections of chlordecone in corn oil at doses of 0.17, 0.34, 1.7, and 3.4 mg/kg for a total of 44 weeks. Controls for this experiment included rats given DEN after partial hepatectomy without chlordecone administration, rats receiving biweekly administration of chlordecone without DEN initiation, and rats receiving corn oil vehicle only. Chlordecone (30 mg/kg) was also administered by corn oil gavage as an initiating chemical given 24 hours after partial hepatectomy. This treatment was followed 10 days later by administration of the tumor promoter sodium phenobarbital in the drinking water at a daily concentration of 0.05% for 44 weeks. A second experiment was conducted that compared promotion in the two-stage assay in male and female rats. A similar study design was used; however, chlordecone was administered biweekly by s.c. injection at higher doses (3 or 9 mg/kg) and the treatment was continued for only 27 weeks.

At the end of each experiment, rats were killed and their livers were evaluated histologically for the presence of preneoplastic lesions (hyperplastic hepatocellular foci) and tumors (hepatocellular carcinomas). Histological staining for GGT was used to identify preneoplastic foci in nontumorous liver sections. Morphometric measurements of GGT-positive foci were determined, and the total number of foci/cm³ of liver were quantified. The concentration of chlordecone in the liver was measured by gas-liquid chromatography.

Body weight gain was not altered in male rats receiving chlordecone at doses between 0.17 and 3.4 mg/kg biweekly for 44 weeks (with or without DEN initiation). Higher doses did affect body weight gain (3 and 9 mg/kg in females and 9 mg/kg only in males) when administered biweekly for 27 weeks. The depression in body weight gain was independent of DEN initiation. Doses greater than 3 mg/kg lead to increased irritability in male and female rats, but no obvious tremors, dermatologic changes, or liver enlargement were observed. Nonneoplastic liver lesions were observed histologically in both male and female rats given chlordecone doses of 3 and 9 mg/kg biweekly (s.c.) for 27 weeks. The lesions included hypertrophy of Zone 3 hepatocytes, congestion, mild fatty change, focal necrosis, and occasional small nests of proliferated sinusoidal cells. The severity of these lesions appeared to be dose-related, although the incidence and severity of noncancer lesions was not quantitatively evaluated.

A dose-related increase in the number of GGT-positive foci/cm³ of liver was observed in male rats given chlordecone at doses between 0.17 and 3.4 mg/kg biweekly (s.c) for 44 weeks following hepatectomy and initiation with DEN (as compared with control groups that were receiving either initiating or promoting treatment alone). Hyperplastic nodules were also observed in 19% of male rats given the initiation and promotion treatments, while nodular liver lesions were not observed in control rats. Chlordecone (30 mg/kg) was not effective as an initiating chemical following partial hepatectomy and promotion with sodium phenobarbital for 44 weeks. A significant sex difference was noted in the chlordecone promotion response at

doses of 3 and 9 mg/kg. Both the median number and the size of the GGT-positive foci were increased in female rats as compared to male rats following DEN initiation and 27 weeks of chlordecone promotion. In addition, hepatocellular carcinomas were observed in female rats (11% at 3 mg/kg and 62% at 9 mg/kg) but were not found in male rats given the same initiation-promotion treatment. Male rats exhibited only preneoplastic foci and nodular hyperplasia under the condition of the two-stage assay. Similar concentrations of chlordecone were measured in the livers of male and female rats, suggesting that enhancement of the tumor promotion response is due to increased sensitivity of females and not altered pharmacokinetics.

Chlordecone was demonstrated to be a liver tumor promoter in a two-stage assay of hepatocarcinogenesis (Sirica et al., 1989). The mode of action for liver tumor promotion by chlordecone is unclear; however, liver toxicity and the subsequent repair/regeneration response may play a role at high doses. Liver toxicity (i.e., focal necrosis, hypertrophy, congestion, and fatty change) and decreased body weight gain were evident in male and female rats at doses that induced liver tumor promotion. However, this study did not evaluate histological evidence of liver toxicity at lower dose levels that were shown to cause an increase in GGT-positive foci in male rats. Therefore, the study did not provide an indication of whether liver toxicity precedes liver tumor promotion (Sirica et al., 1989).

Some *in vitro* evidence suggests that the promotion of liver tumors by chlordecone may be related to suppression of proliferative control through inhibition of gap junctional cell-to-cell communication. The metabolic cooperation between co-cultivated 6-thioguanine-sensitive and resistant Chinese hamster V79 cells was used to evaluate intracellular communication via gap junctions (Tsushimoto et al., 1982). 6-Thioguanine-sensitive cells are wild-type V79 cells that are capable of metabolizing 6-thioguanine to a lethal substrate for nucleic acids that causes cell death. Resistant cells lack the enzyme for 6-thioguanine metabolism; however, cell death can be induced in these cells by a transfer of the lethal 6-thioguanine metabolite across gap junctions from sensitive cells (i.e., metabolic cooperation). Chlordecone was shown to inhibit metabolic cooperation in co-cultivated Chinese hamster V79 cells.

Chlordecone inhibition of cell-to-cell communication was also demonstrated in a dye transfer study in embryonic palatal mesenchymal cells (Caldwell and Loch-Carusio, 1992). Lucifer yellow was scrape-loaded into cell monolayers in the presence or absence of chlordecone. The lucifer yellow dye is too large to cross the plasma membrane but can enter cells through gap junctions. Junctional communication was demonstrated by the movement of lucifer yellow fluorescence away from the scrape line. Chlordecone (20 µg/mL) inhibited dye transfer as demonstrated by the restriction of dye to cells near the scrape line. This effect was reversible with a recovery of dye transfer ability 15 minutes after incubation with control culture medium.

Chlordecone was shown to disrupt adherens junctions in human breast epithelial cells (Starcevic et al., 2001). Human breast epithelial cells cultured on Matrigel (an extracellular

matrix) form lattice-like structures that were disrupted by incubation with 0.1 and 1.0 μM chlordecone (0.01 μM chlordecone had no effect). Chlordecone was also demonstrated to decrease the levels of the transmembrane proteins E-cadherin and β -catenin. These proteins are components of the adherens junctions that mediate cell-to-cell interaction and may play a role in development of neoplastic lesions.

The available data suggest that chlordecone, like many other halogenated hydrocarbons, is not genotoxic, but may act as an epigenetic carcinogen and a tumor promoter. Chlordecone shares similar characteristics with several other well-known tumor promoters. These features include the following: (1) chlordecone induces hepatic enzyme induction (Trosko et al., 1983; Williams, 1980); (2) tumors are found predominantly in rat or mouse livers (NCI, 1976a); (3) chlordecone lacks reactive functional groups and is not genotoxic; (4) there is no evidence of covalent binding to DNA; (5) chlordecone induces ornithine decarboxylase activity (ATSDR, 1995; Mitra et al., 1990; Kitchin and Brown, 1989); and (6) chlordecone inhibits gap-junctional-mediated intercellular communication (Caldwell and Loch-Caruso, 1992; Tsushimoto et al., 1982).

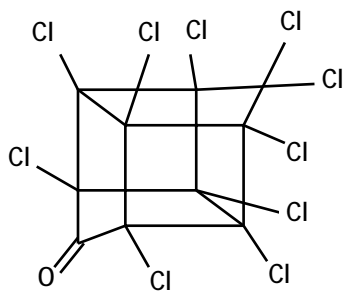
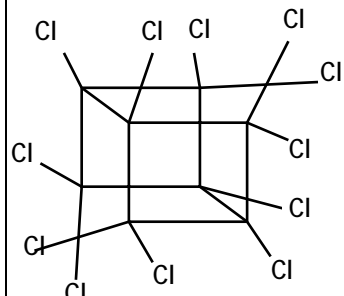
Most of the effects of chlordecone are thought to be produced by the parent compound, primarily by interfering with the function of mitochondrial and cellular membranes. Disruption of cellular homeostasis and energy production within the cell eventually leads to impaired cellular function. In the liver, membrane perturbation and inhibition of transport proteins at the bile canalicular membrane is thought to be related to chlordecone-induced hepatobiliary dysfunction.

4.5.3. Structural Analog Data—Relationship to Mirex

Information on structural analogs can be instructive in predicting biological activity and carcinogenic potential of an agent. Confidence in the conclusions of such a chemical relationship is a function of how similar the analogs are in structure, metabolism, and biological activity. Chlordecone is closely related to the chlorinated pesticide, mirex, in structure, physiochemical properties, and biological activity.

Mirex is a fully chlorinated molecule, whereas chlordecone has a similar structure with only the substitution of two chlorine atoms for a carbonyl group (a double-bonded oxygen atom). This substitution imparts more water solubility as compared to mirex. Both compounds have very low vapor pressures and very high melting points and are crystalline solids at standard conditions. A comparison of physiochemical properties of chlordecone and mirex is presented below in Table 4-17.

Table 4-17. Physiochemical properties of chlordecone and mirex

| | Chlordecone | Mirex |
|-------------------------------------|---|--|
| Structure |  |  |
| Chemical formula | C ₁₀ Cl ₁₀ O | C ₁₀ Cl ₁₂ |
| Molecular weight | 491 g/mol | 546 g/mol |
| Physical state | Crystalline solid | Crystalline solid |
| Octanol-water partition coefficient | 5.41 | 6.89 |
| Water solubility | 2.7 mg/L | 0.085 mg/L |
| Vapor pressure | 2 × 10 ⁻⁷ mm Hg | 8 × 10 ⁻⁷ mm Hg |

Source: NLM (2004b, c).

Mirex and chlordecone are both highly absorbed (75–90%) upon oral exposure and are not substantially metabolized (Egle et al., 1978; Pittman et al., 1976; Wiener et al., 1976). A subset of chlordecone (about 50–75%) is converted into chlordecone alcohol in humans and in some animal species (Fariss et al., 1980; Blanke et al., 1978). No data exist on metabolism of mirex in humans, though animal studies indicate that mirex is not metabolized (Pittman et al., 1976; Wiener et al., 1976; Ivie et al., 1974; Gibson et al., 1972). As a fully chlorinated hydrocarbon, mirex is very hydrophobic and preferentially accumulates in fat (Wiener et al., 1976; Kennedy et al., 1975; Gibson et al., 1972). Chlordecone partitions into fat to a lesser extent. Human data from occupational exposures to chlordecone indicate that chlordecone binds to plasma proteins and lipoproteins and is preferentially sequestered in the liver. The average partitioning of chlordecone among liver, fat, and blood in occupationally exposed workers was found to be 15:7:1 (Cohn et al., 1978).

Chronic exposure studies of chlordecone have indicated that the liver is a target of toxicity. Exposure to chlordecone and mirex in experimental animals results in similar noncancerous liver lesions that may or may not be precursor effects to the development of liver tumors. Liver lesions common to mirex and chlordecone include hypertrophy, hyperplasia, fatty changes, cytoplasmic vacuolation, and anisokaryosis (NTP, 1990; Chu et al., 1981b, c; Larson et al., 1979a, b; NCI, 1976a, b). Though no data exist on liver sensitivity to mirex in humans, observational studies of workers occupationally exposed to chlordecone found evidence of hepatomegaly in 20 workers. Liver biopsies from 12 of these individuals showed histological changes, including proliferation of the SER and cytoplasmic accumulation of lipofuscin (Guzelian, 1982a; Guzelian et al., 1980; Taylor et al., 1978).

Mirex has been shown to induce liver tumors in both sexes of rats and mice in chronic feeding studies at similar dose levels as chlordecone. Incidence of liver tumors in chlordecone-treated male and female rats and mice were found to be significantly elevated at 1.7, 2, 3.4, and 3.5 mg/kg-day. Increased incidence of liver tumors with chronic mirex exposure has been shown in rats and mice at 3.8 and 7 mg/kg-day (NTP, 1990; Innes et al., 1969). In F344/N rats exposed to 0, 0.007, 0.07, 0.7, 1.8, 3.8, and 7.7 mg/kg-day mirex in the diet, statistically significantly increased incidences of combined liver adenomas and carcinomas were found in male and female rats exposed to ≥ 3.8 mg/kg-day mirex (PWG, 1992; NTP, 1990). Incidences for liver adenomas alone were statistically significantly elevated at concentrations ≥ 1.8 mg/kg-day in male and ≥ 3.9 mg/kg-day in female F344/N rats compared with controls. In CD rats exposed chronically in the diet to 0, 4, 7 (males), or 8 (females) mg/kg-day mirex, males showed statistically significantly increased incidences of liver neoplastic nodules and hepatocellular carcinomas at 7 mg/kg-day, whereas females showed increased incidences of neoplastic nodules at 4 and 8 mg/kg-day, with no significant increases in hepatocellular carcinomas at either exposure level (Ulland et al., 1977). In B6C3F₁ and B6AKF₁ mice exposed for life to 0 or 7 mg/kg-day mirex in the diet, liver tumors reported as hepatomas were found at statistically significantly increased incidence in exposed males and females.

Liver tumors resulting from mirex and chlordecone exposure are generally described as well-differentiated masses without vascular invasion or metastases (PWG, 1992; NTP, 1990; Ulland et al., 1977; NCI, 1976a, b; Innes et al., 1969). The available studies on mirex or chlordecone classified liver tumors as either neoplastic nodules or hepatocellular carcinomas (Ulland et al., 1977; NCI, 1976a, b). In vivo and in vitro genotoxicity studies for mirex and chlordecone were generally negative. However, the available evidence for chlordecone and mirex is inadequate to establish a mode of action by which these chemicals induce liver tumors in rats and mice.

Mirex and chlordecone have exhibited similarities in reproductive effects. Decreased sperm counts and testicular degeneration have been observed in animals (Yarbrough et al., 1981; Larson et al., 1979a). Additionally, decreased production of litters in animals was observed for both mirex and chlordecone (Cannon and Kimbrough, 1979; Gaines and Kimbrough, 1970).

It should be noted that although chlordecone and mirex have similar biological activity in the liver at comparable dose levels, some of the observed noncancer effects for these structurally related chemicals are dissimilar. For example, chlordecone exposure results in neurological symptoms, most notably tremors, in experimental animals and in occupationally exposed humans (Taylor, 1985, 1982; Linder et al., 1983; Guzelian, 1982a, b; Larson et al., 1979a; Taylor et al., 1978). However, neurological effects have not been observed with mirex exposure (NTP, 1990; Ulland et al., 1977; Innes et al., 1969). In addition, one of the most sensitive effects of mirex exposure is the development of cataracts in offspring exposed in utero and lactationally, whereas the development of cataracts in offspring does not occur as a result of chlordecone exposure.

Differences in distribution between chlordecone and mirex may contribute to differences in their low-dose biological effects. For instance, it is known that mirex primarily localizes in adipose tissue, whereas chlordecone predominantly accumulates in the liver (Hewitt et al., 1985; Morgan et al., 1979; Cohn et al., 1978; Egle et al., 1978; Wiener et al., 1976; Kennedy et al., 1975).

4.6. SYNTHESIS OF MAJOR NONCANCER EFFECTS

Table 4-18 presents a summary of the noncancer results for the repeated-dose oral studies of chlordecone toxicity in experimental animals. The primary noncancer health effects of occupational exposure to chlordecone in humans and oral exposure in animals include liver lesions, neurotoxicity, and male reproductive toxicity. Kidney effects were also observed in oral exposure studies in animals. Female reproductive effects (i.e., PVE and impaired reproductive success) and developmental effects also occur; however, the doses required to elicit these effects were generally higher than those that resulted in other key effects.

Table 4-18. Summary of noncancer results for oral exposure studies of experimental animals to chlordecone

| Species | Sex | Average daily dose (mg/kg-day) | Exposure duration | NOAEL (mg/kg-day) | LOAEL (mg/kg-day) | Responses | Comments | Reference |
|---|--------|----------------------------------|---|-------------------|-------------------|---|--|--|
| Rat | M F | 0, 0.6, 1.7 0, 1.4, 2.0 | 20 months | ND ND | 0.6 1.4 | Liver histopathology, neurotoxicity | Hyperplasia and tremors; kidney inflammation observed at higher doses | NCI, 1976a |
| Mouse | M F | 0, 3.4, 3.9 0, 3.5, 7.0 | 20 months | ND ND | 3.4 3.5 | Liver histopathology, neurotoxicity | Hyperplasia and tremors | NCI, 1976a |
| Rat | M | 0, 0.07 | 21 months | ND | ND | Liver and thyroid histopathology | No statistically significant increase in incidence due to small number of animals tested and changes in controls | Chu et al., 1981a |
| Rat | M/F | 0, 0.06, 0.3, 0.5, 1.6, 3.9, 7.0 | 2 years | 0.06 | 0.3 | Kidney histopathology | Glomerulosclerosis; higher doses cause fatty changes, hyperplasia in the liver, and tremors | Larson et al., 1979a |
| Dog | M/F | 0, 0.02, 0.1, 0.5 | 128 weeks | 0.1 | 0.5 | Decreased body weight; organ to body weight changes | Magnitude of body weight reduction not reported; small number of animals detract from reliability of study | Larson et al., 1979a |
| Reproductive and Developmental Studies | | | | | | | | |
| Rat | M | 0, 0.625, 1.25, 2.5, 5, 10 | 10 days | ND | 0.625 | Reproductive toxicity | Decreased sperm concentration | U.S. EPA, 1986c |
| Mouse | F | 0, 2, 4, 8 | 4 weeks | ND | 2 | Reproductive toxicity | PVE; higher doses adversely affect follicle size and condition | Swartz and Mall, 1989; Swartz et al., 1988 |
| Rat | M/F | 0, 0.3, 0.5, 1.6, 3.9, 7.0 | 13 weeks | 0.5 | 1.6 | Reproductive toxicity | Testicular atrophy in a subset of animals from the 2-year study | Larson et al., 1979a |
| Rat | M F | 0, 1.4 0, 1.7 | 3 months | 1.4 ND | ND 1.7 | Reproductive toxicity | Impaired reproductive success in females; tremors, liver, and adrenal lesions | Cannon and Kimbrough, 1979 |
| Rat | M | 0, 0.26, 0.83, 1.67 | 3 months | 0.26 | 0.83 | Sperm parameters, neurotoxicity | Decreased sperm motility and viability, hyperexcitability, and mild tremors | Linder et al., 1983 |
| Mouse | M/F | 0, 1.9, 5.6, 7.0 | 1 month prior to mating, 100 days after pairing | 1.9 | 5.6 | Reproductive toxicity | Decrease in the number of pairs producing a second litter; PVE | Huber, 1965 |

Table 4-18. Summary of noncancer results for oral exposure studies of experimental animals to chlordecone

| Species | Sex | Average daily dose (mg/kg-day) | Exposure duration | NOAEL (mg/kg-day) | LOAEL (mg/kg-day) | Responses | Comments | Reference |
|---------|-----|--------------------------------|--|-------------------|-------------------|---|--|---------------------------|
| Mouse | M/F | 0, 0.94 | 1 month prior to mating, 5 months after pairing | ND | 0.94 | Reproductive toxicity | Decrease in the number of pairs producing a second litter | Good et al., 1965 |
| Rat | F | 0, 0.1, 0.6 | 60 days prior to mating and throughout gestation and lactation | 0.1 | 0.6 | Developmental toxicity | WOE for neurobehavioral effects negative; decreased female pup body weight at 100 days | Squibb and Tilson, 1982 |
| Rat | F | 0, 15 | GDs 14–20 | ND | 15 | Reproductive and developmental toxicity | PVE in offspring, decreased ovary weight, increased adrenal weight | Gellert and Wilson, 1979 |
| Rat | F | 0, 2, 6, and 10 | GDs 7–16 | 2 | 6 | Developmental toxicity | Fetotoxicity (decreased fetal body weight); maternal toxicity at lower doses | Chernoff and Rogers, 1976 |
| Mouse | F | 0, 2, 4, 8, and 12 | GDs 7–16 | 8 | 12 | Developmental toxicity | Fetotoxicity (fetal mortality); maternal toxicity at lower doses | Chernoff and Rogers, 1976 |

ND = not determined; WOE= weight of evidence

4.6.1. Oral

Liver enlargement developed in 20 out of 32 workers exposed to high levels of chlordecone for an intermediate to chronic exposure duration; however, evidence of significant liver toxicity was not found (Guzelian, 1982a; Guzelian et al., 1980; Taylor et al., 1978). Normal results were obtained for serum biochemistry, and liver biopsy samples showed histological changes in the liver that were characterized as nonadverse in nature by study authors (see Section 4.1). Histological changes included proliferation of the SER and cytoplasmic accumulation of lipofuscin. No evidence of fibrosis, cholestasis, or hepatocellular necrosis was found; however, the exposure duration and latency period before examination were relatively short.

Histological changes in the liver have also been demonstrated in laboratory animals. These effects include increased liver size and weight, hepatocellular hypertrophy, proliferation of the SER, increased microsomal protein, CYP450 content, cytochrome c reductase activity, and microsomal enzyme activity (see Section 3.3) (Gilroy et al., 1994; Hewitt et al., 1985; Mehendale et al., 1978, 1977). Histopathological evidence of hepatotoxicity was also demonstrated in animals following chronic exposure to chlordecone. The liver lesions observed in male and female rats given chlordecone doses of 3 and 9 mg/kg biweekly (s.c.) for 27 weeks (average daily doses of 0.86 and 2.6 mg/kg-day) included hepatocellular hypertrophy, congestion, mild fatty change, focal necrosis, and occasional small nests of proliferated sinusoidal cells (Sirica et al., 1989). Fatty changes and hyperplasia were also observed in rats given doses >0.5 mg/kg-day for up to 2 years (Larson et al., 1979a).

Kidney toxicity was reported in laboratory animals, but was not observed in occupationally exposed pesticide workers (Taylor, 1985, 1982; Guzelian, 1982a, b; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). It is possible that the clinical signs of glomerulosclerosis (including proteinuria) were not observed in occupationally exposed pesticide workers because of the relatively short exposure duration (average exposure duration was 5–6 months), which may not be a sufficient duration for the development of more obvious renal disease (nephropathy and frank proteinuria). It is unclear whether clinical tests sufficient to detect glomerular damage were performed on the exposed workers. Furthermore, a definitive diagnosis of glomerulosclerosis can only be diagnosed through a kidney biopsy, which was not performed on any occupationally exposed worker. Larson et al. (1979a) identified a chronic LOAEL of 0.3 mg/kg-day for proteinuria and increased incidence of glomerulosclerosis in female Wistar rats with a corresponding NOAEL of 0.06 mg/kg-day. Renal effects were also reported in other studies at higher dose levels. NCI (1976b) included summary tables in which chronic kidney inflammation in male Osborne-Mendel rats (at 0.6 mg/kg-day) and female Osborne-Mendel rats (at 2.0 mg/kg-day) was reported. Chu et al. (1980) reported that 28 days of dietary exposure to chlordecone (at

0.07 mg/kg-day) produced eosinophilic inclusions in proximal tubules in 2/10 male Sprague-Dawley rats.

Neurological symptoms, including tremor, headache, and irritability, were reported in workers exposed to high doses of chlordecone for a period of months to years (see Section 4.1) (ATSDR, 1995; Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). Nearly half (7/16) of the workers reported persistent symptoms (e.g., tremor, nervousness) 5 to 7 years later (Taylor, 1985). In laboratory animals, chlordecone has been shown to cause tremors, decreased motor coordination, hyperexcitability, and an exaggerated startle response (Linder et al., 1983; Huang et al., 1980; Larson et al., 1979a; NCI, 1976a). The hypothesized mode of action for neurotoxicity relates to alteration in membrane transport proteins and disruption of calcium homeostasis (see Section 4.4.3). In the chronic rat study by Larson et al. (1979a), liver lesions were observed at slightly lower doses (≥ 0.5 mg/kg-day) than those resulting in clinically observable tremors (≥ 1.6 mg/kg-day); however, hyperexcitability and mild tremors were observed in a subchronic dietary study in rats at doses as low as 0.83 mg/kg-day (Linder et al., 1983).

Chlordecone exposure in humans caused oligospermia, reduced sperm motility, and decreased libido in a group of men who were occupationally exposed to chlordecone for periods of up to 1.5 years (see Section 4.1) (Taylor, 1985, 1982; Guzelian, 1982a; Taylor et al., 1978). Upon follow up 5 to 7 years following the cessation of chlordecone exposure and treatment with cholestyramine, male reproductive parameters had returned to normal (Taylor, 1982). Even though two of seven workers sired children, there is no indication of the true denominator of how many were trying to conceive and/or the fertility rate. Chlordecone-induced male reproductive toxicity has also been observed in laboratory animal studies (Linder et al., 1983; Larson et al., 1979a). Sperm parameters were altered by chlordecone in a subchronic dietary study (Linder et al., 1983). Sperm viability, motility, and reserves in the right cauda epididymis were significantly reduced at doses of 0.83 and 1.67 mg/kg-day but not at 0.26 mg/kg-day. Reproductive performance (determined by number of pregnant females, number of live litters, average live litter size, number of implants, percentage of resorptions, and fetal weight) was similar across exposed and control groups. No gross or microscopic pathology of the male reproductive system was found that could be attributed to chlordecone treatment, and recovery from the reported sperm alterations was apparent 4.5 months following cessation of exposure. Decreased sperm concentration was observed in rats exposed to chlordecone doses ≥ 0.625 mg/kg-day for 10 days (U.S. EPA, 1986c). Testicular atrophy was observed in rats at doses ≥ 1.6 mg/kg-day for 13 weeks (Larson et al., 1979a).

No information is available concerning chlordecone-induced reproductive effects in women. Impaired reproductive success was, however, observed in mice and rats exposed to chlordecone at doses of ≥ 1 mg/kg-day (see Section 4.3.1) (Cannon and Kimbrough, 1979; Good

et al., 1965; Huber, 1965). The mechanism responsible for impaired reproductive success is unknown; however, chlordecone has been demonstrated to affect estrous cyclicity in female mice (Swartz and Mall, 1989; Swartz et al., 1988; Huber, 1965). Huber (1965) demonstrated that PVE occurs within 8 weeks of chlordecone treatment at doses of ≥ 5.6 mg/kg-day. Similar effects on estrous cyclicity were noted by Swartz and Mall (1989) and Swartz et al. (1988) within 2 weeks of chlordecone administration at dose levels of 2, 4, and 8 mg/kg-day. After 4 and 6 weeks of treatment, ovulation was reduced in the highest chlordecone treatment group (8 mg/kg-day), which resulted in statistically significantly lower numbers of ovulated oocytes relative to vehicle controls (Swartz et al., 1988). PVE was also observed in offspring of female rats given 15 mg/kg-day chlordecone by gavage on GDs 14–20 (Gellert and Wilson, 1979). Female offspring also exhibited significantly decreased ovarian weight, significantly increased adrenal weight (relative to vehicle controls), and a decrease in the number of animals ovulating.

No information is available concerning developmental effects of chlordecone exposure in humans. Laboratory animal studies demonstrated developmental toxicity in rats and mice at dose levels that also produced maternal toxicity (Chernoff and Rogers, 1976). Chernoff and Rogers (1976) demonstrated that chlordecone administration via gavage during GDs 7–16 induced maternal toxicity in mice and rats at doses ≥ 2 mg/kg-day, while fetotoxicity did not occur until doses of ≥ 6 mg/kg-day in rats and ≥ 12 mg/kg-day in mice. Maternal toxicity was evidenced by decreased body weight and increased liver to body weight ratios. Fetotoxicity in rats was observed as significantly depressed fetal body weight and delayed ossification in 6 and 10 mg/kg-day dose groups and significantly increased incidences of fetuses with enlarged renal pelvis, edema, undescended testes, or enlarged cerebral ventricles in the 10 mg/kg-day group relative to controls. Signs of fetotoxicity in mice were observed only in the highest dose group and consisted of significantly increased fetal mortality.

The mode of action of chlordecone-induced toxicity is not completely understood. However, limited evidence suggests that chlordecone may interact with cell membranes and affect the membrane transport proteins (e.g., Mg^{2+} -ATPase, Ca^{2+} -ATPase) that are responsible for cellular homeostasis and energetics. Disruption of cellular homeostasis and energy production within the cell leads to impaired cellular function. In the central nervous system, altered calcium homeostasis leads to changes in neurotransmitter activity (e.g., alpha-noradrenergic, dopaminergic, and serotonergic systems) that may be related to chlordecone-induced tremor and exaggerated startle response (Vaccari and Saba, 1995; Brown et al., 1991; Herr et al., 1987; Desaiyah, 1985; Fujimori et al., 1982b; Squibb and Tilson, 1982). In the liver, membrane perturbation and inhibition of the active transport of glutamate at the bile canalicular membrane may be related to chlordecone-induced hepatobiliary dysfunction (Teo and Vore, 1991). Additionally, chlordecone alters calcium homeostasis in hepatocytes, leading to a decline in glycogen storage and a reduced energy status (Kodavanti et al., 1993, 1990).

An estrogenic mode of action is generally considered to be involved in the reproductive toxicity of chlordecone. Testicular atrophy, altered sperm characteristics, persistent vaginal estrus, and anovulation observed in chlordecone-treated laboratory animals (Swartz et al., 1988; U.S. EPA, 1986c; Linder et al., 1983; Larson et al., 1979a; Huber, 1965) mimic the effects produced by excessive estrogen. Mechanistic studies demonstrate that chlordecone binds to the estrogen receptor, as well as other endocrine receptors (see Section 4.4.4).

4.6.2. Mode-of-Action Information—Glomerular Lesions

The mechanism by which chronic dietary chlordecone exposure in rats results in glomerular lesions is unclear. Larson et al., 1979a observed a significant, dose related increase in the incidence and severity of renal lesions in female Wistar rats in the 0.3, 0.5, and 1.6 mg/kg-day dose groups. An increase in proteinuria, a clinical sign of glomerular damage, was also observed in female rats, starting at 0.3 mg/kg-day (see also Section 4.2.2.1).

The Larson (1979a) study itself does not inform the potential mode of action of the observed glomerular lesions; however, there are some data to suggest that the effect may be mediated through an autoimmune mechanism. Glomerular damage is often, though not exclusively, mediated through immune mechanisms (U.S. DHHS, 2006). Some evidence (Sobel et al., 2006, 2005) suggests that chlordecone may accelerate glomerular lesions in susceptible animals by way of increased deposition of immune complexes in the glomeruli (see Section 4.4.5). In similar treatment protocols Sobel et al. (2006, 2005) implanted female (NZB × NZW)F₁ mice with sustained-release pellets containing 0.001, 0.01, 0.1, 0.5, 1, or 5 mg chlordecone s.c. above the shoulders. Ovary intact mice treated with either 1 mg or 5 mg chlordecone pellets developed anti-dsDNA and antichromatin autoantibody titers significantly earlier than controls. Additionally, immunohistofluorescence analysis of renal sections from a subset of animals treated for 8 weeks with 1 mg chlordecone showed enhanced deposits of IgG immune complexes as compared with untreated controls. The histopathology associated with renal disease was similar between chlordecone-treated mice and controls.

An alternate theory holds that chlordecone damages the glomeruli directly. Chlordecone predominantly binds plasma proteins and lipoproteins (especially albumin and HDL); this binding has been demonstrated in exposed workers and in animal models (Soine et al., 1982; Skalsky et al., 1979). The glomeruli are the functional units of the kidney that are predominantly responsible for filtering high molecular weight proteins, including albumin, from the blood (Hart and Kinter, 2005). Therefore, this region of the kidney may be subjected to relatively high concentrations of chlordecone that could potentially result in direct chemical insult. Distribution studies in experimental animals by various routes of exposure (see Section 3.2) have indicated that chlordecone predominantly localizes in the liver, but is also distributed to the kidneys (Belfiore et al., 2007; Heatherington et al., 1998; Hewitt et al., 1985; Kavlock et al., 1980). A

dermal study of chlordecone organ distribution found that kidney concentration was second only to liver concentration (Heatherington et al., 1998).

Uncertainty surrounds the mechanism for the observed glomerular damage following chlordecone exposure. It is conceivable that chlordecone may not *cause* glomerular damage per se but that it may accelerate or increase the severity of the disease in animals with preexisting susceptibility to glomerular damage. For example, though a significant dose-response relationship was seen in the principal study between glomerulosclerosis and increasing doses of chlordecone, the control animals also exhibited a background incidence of glomerular lesions, which was particularly high in the male rats (12% incidence in females and 55% in males). In addition, Sobel et al. (2006, 2005) indicated that chlordecone exposure increased the severity and accelerated the development of renal damage and autoantibodies in a susceptible mouse strain, (NZB × NZW)_{F1}. However, a follow-up experiment by Sobel et al. (2006) treated BALB/c mice, a strain in which spontaneous development of glomerular damage is rare, and found that treatment of these mice with chlordecone for up to 1 year did not produce elevated autoantibody titers or renal disease.

4.7. EVALUATION OF CARCINOGENICITY

4.7.1. Summary of Overall Weight of Evidence

Under the U.S. EPA *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), chlordecone is “likely to be carcinogenic to humans” based on data from an oral cancer bioassay in rats and mice demonstrating an increase in the incidence of hepatocellular carcinomas in both sexes of both species (NCI, 1976a, b). NCI (1976a, b) reported a statistically significant increase in hepatocellular carcinomas in both sexes of mice. Male and female rats exhibited increased incidences of hepatocellular carcinomas at high doses that were statistically significant when compared with pooled controls. The incidence of hepatocellular carcinomas was not statistically significant in comparison with matched controls for rats of either sex. The tumor response was particularly robust in male and female mice at the highest doses (Table 4-1). NCI (1976a, b) also demonstrated a decrease in the time to tumor in both sexes of both species. No other tumor types were significantly increased in either rats or mice in this study.

There are no studies in humans that assess the carcinogenic potential of chlordecone. Other chronic animal studies of chlordecone (Chu et al., 1981a; Larson et al., 1976a) lacked adequate power to detect carcinogenicity. Chu et al. (1981a) included only one dose group of 10 animals/sex and did not use an adequately high dose (0.07 mg/kg-day). The study by Larson et al. (1979a) also was limited in power. Only four animals/sex were examined in the highest dose group (1.6 mg/kg-day) at the termination of the study.

Similarities in the tumor profile of chlordecone and mirex, a structurally related chemical, have been observed in animals. Mirex has been shown to induce hepatocellular adenomas or carcinomas in both sexes of rats and mice (PWG, 1992; NTP, 1990; Ulland et al., 1977; NCI,

1976a, b; Innes et al., 1969). A statistically significantly increased incidence of liver tumors in F344/N and CD rats and B6C3F₁ and B6AKF₁ mice has been observed following chronic oral exposure to mirex at similar dose levels as chlordecone. The liver tumors resulting from exposure to mirex, similar to exposure to chlordecone, are described as predominantly well-differentiated masses without vascular invasion or metastases (PWG, 1992; NTP, 1990; Ulland et al., 1977; NCI, 1976a, b; Innes et al., 1969). Mirex and chlordecone also produce noncancer effects in the liver at similar doses. It should be noted that, though chlordecone and mirex appear to have closely related biological activity and carcinogenicity in the liver at similar dose levels (though the mode of action for each is unknown), several noncancer effects reported following exposure to mirex and chlordecone are dissimilar. For example, the characteristic neurotoxicity observed following exposure to chlordecone has not been described for mirex.

The mode of carcinogenic action of chlordecone in the livers of rats and mice is unknown. Most genotoxicity tests for chlordecone are negative. For the liver tumors in rats and mice, some data suggest that chlordecone may induce cell proliferation and lead to a promotion in the growth of preinitiated cells. However, key precursor events linked to observed cell proliferation have not been identified.

U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a) indicate that for tumors occurring at a site other than the initial point of contact the weight of evidence for carcinogenic potential may apply to all routes of exposure that have not been adequately tested at sufficient doses. An exception occurs when there is convincing toxicokinetic data that absorption does not occur by other routes. For chlordecone, systemic tumors were observed in rats and mice following oral exposure. No animal cancer bioassay data following inhalation or dermal exposure to chlordecone are available. Data evaluating absorption by the inhalation route are unavailable and limited data are reported for dermal absorption (Heatherington et al., 1998; Shah et al., 1987). However, based on the observance of systemic tumors following oral exposure, and in the absence of information to indicate otherwise, it is assumed that an internal dose will be achieved regardless of the route of exposure. Therefore, chlordecone is "likely to be carcinogenic to humans" by all routes of exposure.

4.7.2. Synthesis of Human, Animal, and Other Supporting Evidence

Few studies are available that directly assess the carcinogenic potential of chlordecone. Limited data on the carcinogenic potential in humans can be garnered from observational studies of a single group of 133 workers occupationally exposed to chlordecone at a chlordecone manufacturing plant in Hopewell, Virginia, in the late 1970s (Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). A subset of 32 of these workers with clinical signs or symptoms of chlordecone toxicity and high chlordecone blood levels (>0.6 µg/mL at the time of diagnosis) were examined specifically for hepatotoxicity (Guzelian et al., 1980). Hepatomegaly was

observed in 20/30 of these workers. However, liver biopsy samples taken from 12 of these workers showed no evidence of liver hyperplasia (Guzelian, 1982a; Guzelian et al., 1980). The average exposure duration of these subjects was 5–6 months, and they were physically examined for this study within 10 months of exposure cessation. Upon follow-up of the exposed workers 2–3 years after exposure cessation, hepatomegaly had resolved in all workers and biopsies were negative for abnormal histopathological findings (Guzelian et al., 1980). Conclusions regarding cancer from this study are limited by the small number of workers examined, uncertainties concerning exposure dose and route, the relatively brief duration of exposures, and the absence of a sufficient latency period for tumor development.

Occupational exposures to chlordecone also provide evidence for the preferential accumulation of chlordecone in the liver. For example, in 32 workers exposed to chlordecone for a period that ranged from 3 to 16 months, high concentrations of chlordecone were found in blood, liver, and s.c. fat (Cohn et al., 1978). The ratio of the chlordecone concentration in fat as compared to the chlordecone concentration in the blood was 7:1. However, the liver to blood concentration ratio in exposed workers was reported to be 15:1 (Table 3-2). Chlordecone has also been shown to bind plasma proteins and lipoproteins and preferentially accumulate in the liver, where it is slowly eliminated, in experimental animals and exposed workers (Cohn et al., 1978; Egle et al., 1978). Thus, due to the preferential accumulation of chlordecone in the liver, humans may be susceptible to chlordecone-induced liver toxicity.

The human case reports and clinical observations of occupational chlordecone exposure lack sufficient design, power, and follow-up to determine carcinogenic potential of chlordecone in humans; however, the observations from these studies provide valuable information on human susceptibility to chlordecone. A review of biological and epidemiological evidence of cancer found no population-based studies on cancer in humans related to chlordecone exposure (Ahlborg et al., 1995).

Animal studies provide evidence for the carcinogenic potential of chlordecone. Chlordecone has been shown to induce liver tumors in Osborne-Mendel rats and B6C3F₁ mice in a study performed by the NCI (NCI, 1976a, b). B6C3F₁ mice (50/sex/group) and Osborne-Mendel rats (50/sex/group) were exposed to chlordecone in the diet for 20 months. Dietary concentrations of chlordecone began at 0, 15, 30, or 60 ppm for male rats and 0, 30, or 60 ppm for female rats. In mice, dietary concentrations of chlordecone began at 0 or 40 ppm (two groups at this concentration) for males and 0, 40, or 80 ppm for females. During the course of the study, concentrations were reduced at least once in each treatment group due to toxicity (see Figures 4-1 to 4-4). Time-weighted-average dietary concentrations reported by the study authors were 0, 8, or 24 ppm (0, 0.6, or 1.7 mg/kg-day) for male rats and 0, 18, or 26 ppm (0, 1.4, or 2.0 mg/kg-day) for female rats. In mice, time-weighted-average dietary concentrations were 0, 20, or 23 ppm (0, 3.4, or 3.9 mg/kg-day) for male mice and 0, 20, or 40 ppm (0, 3.5, or 7.0 mg/kg-day) for female mice. Liver tumors described as hepatocellular carcinomas were observed in high-dose

female rats at an incidence that was significantly elevated compared with the pooled control incidence (0/100, 0/10, 1/49, and 10/45 in the pooled control, matched control, low-dose and high-dose groups, respectively). Incidences of male rats with hepatocellular carcinomas were lower at 0/105, 0/10, 1/50, and 3/44, respectively. The incidence of carcinomas in high-dose males was statistically significant ($p = 0.049$) in comparison with pooled controls. The incidence of hepatocellular carcinomas was not statistically significant in comparison with matched controls ($n = 10$) for rats of either sex. A significant dose-response trend was observed for the incidence of hepatocellular carcinoma in both male and female rats (Cochran-Armitage test conducted for this review). In mice, statistically significant elevated incidences of hepatocellular carcinomas were found in both exposed groups compared with matched and pooled control incidences (NCI, 1976a). Incidences for matched control, low-, and high-dose groups were 6/19, 39/48, and 43/49 for male mice and 0/10, 26/50, and 23/49 for female mice. No other tumor types in rats or mice were found to be significantly elevated in this study.

Decreases in survival rates and decreased body weight gain were observed in all animal groups except the low- and high-dose female mice (see Table 4-4). A robust liver tumor incidence of 26/50 (52%) was observed in the low-dose group (3.5 mg/kg-day) of female mice, a group that had survival rates and body weight gains that were comparable with controls. While it is true that high toxicity was observed in the high-dose groups (specifically of male rats and mice), the conclusion that high toxicity is required for tumor induction may not be warranted.

The primary limitation of the NCI (1976a, b) bioassay relates to the dose selection. The initial dietary concentrations in the high-dose groups were excessively high and induced early mortality, tremors, anemia, and dermatitis in both sexes of both species. During the course of the study, concentrations were reduced at least once in each treatment group due to overt toxicity (see Figures 4-1 to 4-4). In both male rats and mice, the initial high-dose group was discontinued due to excessive toxicity and mortality (animals were sacrificed). Because of changes in chlordecone dietary exposure levels, the dose metric related to the development of liver tumors is uncertain.

Conclusions from cancer bioassays utilizing potentially excessive doses are regarded with caution for several reasons. Doses of an agent that cause high toxicity to the animals may result in early deaths directly resulting from toxicity, which could decrease the ability of the assay to detect tumor effects. Animal mortality in the NCI (1976a, b) study was high in comparison to controls; however, this did not prevent the detection of elevated rates of hepatocellular carcinoma in the high-dose groups. Alternately, there is concern that high doses may result in tumor effects that are secondary to toxic effects (e.g., cytotoxicity) or altered toxicokinetics (U.S. EPA, 2005a). It is possible that high doses of chlordecone used in the NCI study resulted in tumor effects that were secondary to liver cytotoxicity and thus would not be likely at low doses. However, there is not sufficient data to support this mode of action. In the absence of data that indicate that direct liver cytotoxicity at high doses precedes tumor development, the increased

incidence of liver tumors observed in the NCI cancer bioassay cannot be discounted. There are no data to support the concern that elevated levels of hepatocellular carcinoma detected by the NCI study in rats and mice are the direct result of altered toxicokinetics from excessive chlordecone levels. In fact, animal data support the conclusion that the liver is especially sensitive to chlordecone-induced lesions even at very low doses that do not result in overt toxicity to the animal or decreased survival (Chu et al., 1981a; Larson et al., 1979a). Additionally, chlordecone has been demonstrated in humans and animals to preferentially accumulate in the liver (Hewitt et al., 1985; Cohn et al., 1978; Egle et al., 1978). Therefore it is not likely that liver tumors arising after high exposures to chlordecone are due to altered toxicokinetics.

Besides the NCI (1976a, b) cancer bioassay, Larson et al. (1979a) and Chu et al. (1981a) are the only additional chronic dietary studies of chlordecone exposure in animals. Larson et al. (1979a) fed groups of Wistar rats (40/sex/group) diets estimated to result in dose levels of 0, 0.06, 0.3, 0.5, 1.6, 3.9, or 7.0 mg/kg-day for up to 2 years. Increased incidence of liver lesions (characterized as fatty changes and hyperplasia) were seen in females at 0.5 mg/kg-day and in males at 1.6 mg/kg-day. Liver lesions in three females in the 0.5 mg/kg-day group and one female and two males in the 1.6 mg/kg-day group were described by the authors as being possibly carcinomatous in nature, though the authors reported that an independent review by four pathologists was equivocal. However, it should be noted that very few animals were available for pathological examination at the end of the study, limiting the study's power to detect carcinogenic effects.

A 21-month dietary exposure study by Chu et al. (1981a) detected an increase in liver lesions in rats in the single chlordecone exposure group (5/6 compared to 3/7) of 0.07 mg/kg-day but did not report tumors. However, the very small number of animals and the use of only a single low-dose group severely limit this study's power to assess carcinogenic potential. Additionally, neither toxicity nor changes in body weight gain were observed in the dose tested. Therefore, the dose utilized cannot be considered adequately high to detect carcinogenic potential for chlordecone.

The structurally related chemical mirex has been shown to induce liver tumors in both sexes of rats and mice in chronic feeding studies at similar dose levels as chlordecone. Incidences of liver tumors in chlordecone-treated male and female rats and mice were found to be significantly elevated at 1.7, 2, 3.4, and 3.5 mg/kg-day. Increased incidence of liver tumors (adenomas or carcinomas) with chronic mirex exposure has been shown in rats and mice at 3.8 and 7 mg/kg-day (PWG, 1992; NTP, 1990; Ulland et al., 1977; Innes et al., 1969). Liver tumors resulting from mirex and chlordecone exposure are generally described as well-differentiated masses without vascular invasion or metastases (PWG, 1992; NTP, 1990; Ulland et al., 1977; NCI, 1976a, b; Innes et al., 1969). The available studies on mirex or chlordecone classified liver tumors as either neoplastic nodules or hepatocellular carcinomas (Ulland et al., 1977; NCI,

1976a, b). In vivo and in vitro genotoxicity studies for mirex and chlordecone were generally negative. However, the available evidence for chlordecone and mirex is inadequate to establish a mode of action by which these chemicals induce liver tumors in rats and mice. Chlordecone and mirex exposure in experimental animals results in similar noncancerous liver lesions that may be precursor lesions to the development of liver tumors. Liver lesions common to mirex and chlordecone include hypertrophy, hyperplasia, fatty changes, cytoplasmic vacuolation, and anisokaryosis (NTP, 1990; Chu et al., 1981b, c; Larson et al., 1979a, b; NCI, 1976a, b). However, though chlordecone and mirex appear to have related biological activity and carcinogenicity in the liver, this evidence is limited by the observation of several dissimilar noncancer effects.

In summary, under U.S. EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), chlordecone is *likely to be carcinogenic to humans*. This characterization is based on a statistically significant increase in liver tumors in both sexes of rats and mice that was observed following oral exposure to chlordecone in a cancer bioassay by the NCI (NCI, 1976a, b).

4.7.3. Mode-of-Action Information

The majority of studies on chlordecone were negative for genotoxic activity in a variety of short-term in vitro and in vivo assays (see Section 4.4.1). One hypothesis for the mode of action of chlordecone induced tumorigenicity is sustained proliferation of spontaneously transformed liver cells, resulting in the eventual formation of liver tumors. Proliferative liver lesions (hyperplasia) were found in a chronic dietary study in Wistar rats at doses greater than 0.5 mg/kg-day in females and 1.6 mg/kg-day in males (Larson et al., 1979a). Additionally, the NCI (1976a, b) chronic dietary cancer bioassay that reported increased incidences of liver tumors in both sexes of rats and mice also noted extensive liver hyperplasia in both sexes of both species. Though the incidence of hyperplasia was not noted in the study, the authors reported that the incidence of hyperplasia in the matched control mice was low as compared to the treated groups. In rats, the authors reported that no liver hyperplasia was seen in the matched controls. Chlordecone was demonstrated to be a liver tumor promoter, rather than an initiator or a complete hepatic carcinogen, in a two-stage tumor promotion assay in male and female Sprague-Dawley rats (Sirica et al., 1989). This study also demonstrated a greater tumor response in female rats, suggesting that hormonal involvement may be important in the promotion of chlordecone-induced liver tumors. The NCI (1976a, b) study provides further support for this potential mode of action for chlordecone. The authors reported an increased incidence of liver tumors and shorter time to tumor formation in female rats exposed to the high dose compared to male rats exposed to the high dose (NCI, 1976a).

Chlordecone is one of a large number of organochlorine chemicals that produce liver tumors in rodents and do not exhibit genotoxicity in short-term tests. Many of these pesticides (including chlordane, heptachlor, and hexachlorocyclohexane) have been shown to promote liver

tumors in rodent livers when administered after an initiating dose of a known carcinogen (Deml and Oesterle, 1987; Williams and Numoto, 1984; Williams, 1983). However, the mode of action by which chlordecone produces liver tumors is unknown. Precursor events in which chlordecone may promote proliferation of transformed liver cells are uncertain, and data regarding a plausible temporal progression from chlordecone-induced liver lesions to eventual liver tumor formation are not available. Therefore, the available evidence is inadequate to establish a mode of action by which chlordecone induces liver tumors in rats and mice.

4.8. SUSCEPTIBLE POPULATIONS AND LIFE STAGES

4.8.1. Possible Childhood Susceptibility

Neurological studies suggest that the immature brain may be sensitive to subtle effects from chlordecone exposure. As reported in Section 4.3, exposure of female rats to chlordecone for 60 days prior to mating through lactation day 12 produced subtle neurological changes in male offspring (Squibb and Tilson, 1982). Behavioral testing of offspring was primarily negative. The only neurobehavioral endpoint detected was a significant increase in the time required to reorient to a vertical position in an assay for negative geotaxis in male offspring of dams exposed to 0.6 mg/kg-day at 100 days of age. In addition, motor activity induced by a subcutaneously injected dopamine receptor agonist was significantly increased in male offspring compared to controls. This suggests an alteration in dopamine sensitivity in male offspring. However, the biological significance of this effect is unclear as spontaneous motor activity of exposed offspring in the absence of pharmacological challenges was not different from controls.

In a lactation exposure study, Sprague-Dawley rat pups were exposed to chlordecone in milk by treating lactating dams immediately after birth with 0 (corn oil vehicle) or 2.5 mg/kg-day chlordecone by gavage (Jinna et al., 1989). In vitro assays of brain P2 fractions showed that the exposed pups (through day 20) exhibited increased activity of Na⁺, K⁺, and Ca⁺²-ATPase activity. As compared to effective doses in adult rats (8.3 mg/kg-day orally for 3 days; Kodavanti et al., 1990), the exposure doses expected via lactation are lower, suggesting that the maturing ATPases of neonatal rats may be more sensitive to chlordecone exposure. At the cellular level, Hoskins and Ho (1982) also reported significant differences in calcium content and subcellular distribution in brain in adult (24 weeks old) as compared to young (4–6 weeks old) male ICR mice following acute oral chlordecone exposure (25 mg/kg-day in corn oil).

In summary, some studies have indicated that developing animals may be more susceptible to subtle neurological effects of chlordecone including alterations in orientation reflex, dopamine sensitivity, ATP-ase activity, calcium concentration and subcellular distribution. Data to inform potential early life susceptibility of other effects of chlordecone are lacking and thus present an additional area of uncertainty.

4.8.2. Possible Gender Differences

The extent to which men and women differ in susceptibility to chlordecone toxicity is not known. No human data are available to suggest that there are gender differences in the toxicity or carcinogenicity of chlordecone.

In the NCI (1976a) bioassay of chlordecone carcinogenicity, a strong liver tumor response was seen in female rats, and only a weak response was noted among male rats. Tumors were seen in both genders of mice; however, mortality in female mice at high doses was lower compared to males. A significant sex difference was noted in the liver tumor promotion response in a two-stage assay of hepatocarcinogenesis (Sirica et al., 1989). Both the median number and the size of the GGT-positive foci were increased in female rats as compared to males following DEN initiation and 27 weeks of chlordecone promotion. In addition, hepatocellular carcinomas were observed in female rats but were not found in male rats given the same initiation-promotion treatment. Similar concentrations of chlordecone were measured in the livers of male and female rats, suggesting that enhancement of the tumor promotion response is due to increased sensitivity of females and not altered pharmacokinetics. It is possible that the estrogenic properties of chlordecone may play a role in the sensitivity of female rats to tumor promotion. Female rats in this study were also more susceptible to decreases in body weight gain, suggesting that enhanced toxicity may play a role in tumor promotion; however, histological examination of noncancerous portions of the liver did not indicate significant gender differences in liver toxicity.

Chlordecone induces reproductive effects in both male and female laboratory animals. However, some evidence exists to suggest that female reproductive toxicity has a larger effect on reproductive success at the same chlordecone dose level. Reproductive toxicity has been demonstrated by altered sperm parameters, testicular atrophy, altered estrous cyclicity, and impaired reproductive success in animals. Although the most sensitive endpoint evaluated appeared to be alterations in sperm parameters induced by subchronic chlordecone exposure in male rats (Linder et al., 1983), these decreases were observed at doses where reproductive success was unaffected. A crossover study in rats that paired control males with treated females and control females with treated males suggests that female reproductive toxicity had a larger effect on reproductive success at the same chlordecone dose level (Cannon and Kimbrough, 1979). In male and female rats fed diets containing 25 ppm chlordecone (1.4 or 1.7 mg/kg-day) for 3 months, 12/20 pairs of treated males and control females produced offspring, while none of the 20 pairs of treated females and control males produced offspring.

5. DOSE-RESPONSE ASSESSMENTS

5.1. ORAL REFERENCE DOSE (RfD)

5.1.1. Choice of Principal Study and Critical Effect—with Rationale and Justification

The only available data concerning health effects of chlordecone in humans are derived from studies of a single group of 133 men exposed occupationally to chlordecone in the late 1970s at a chlordecone manufacturing facility in Hopewell, Virginia (Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). Due to inadequate industrial safety measures at the factory, substantial inhalation, dermal, and oral exposures likely occurred (Cannon et al., 1978). Toxicity observed in the exposed workers included effects on the nervous system, liver, and reproductive system. Of the 133 men, 76 experienced neurological symptoms, especially tremors, nervousness, and headaches, sometimes persisting for as long as 9–10 months after cessation of exposure and the start of treatment (Cannon et al., 1978). In addition, a subset of the men experienced reproductive effects, including oligospermia, reduced sperm motility, and decreased libido (Taylor, 1982). A subset of 32 of the occupationally exposed workers with clinical signs or symptoms of chlordecone toxicity and high chlordecone blood levels ($>0.6 \mu\text{g/mL}$ at the time of diagnosis) were examined specifically for hepatotoxicity (Guzelian et al., 1980). Hepatomegaly was observed in 20/32 workers. Minimal elevation (less than twofold) of ALP was noted in seven patients; however, other liver enzymes were normal including ALT, AST, and GGT (Guzelian et al., 1980). Sulfobromophthalein retention, a measure of liver clearance, was normal in a subset of 18 workers tested (Guzelian et al., 1980). Upon biopsy of 12 workers with hepatomegaly, histological changes included proliferation of the SER and cytoplasmic accumulation of lipofuscin. These changes in the liver were characterized by the authors as nonadverse in nature and were suggested to be adaptive changes rather than a reflection of hepatotoxicity (Guzelian, 1982a, b; Guzelian et al., 1980; Taylor et al., 1978). Upon follow-up of the exposed workers 2–3 years after exposure cessation, hepatomegaly had resolved in all workers and biopsies were negative for abnormal histopathological findings (Guzelian et al., 1980).

Because of uncertainties regarding exposure routes and exposure levels at the facility, NOAELs or LOAELs could not be established for the observed neurological, liver, and reproductive effects in the occupationally exposed workers. Additionally, workers may have had concomitant exposure to the chemical precursors used to manufacture chlordecone. Because of these major uncertainties, health effects data in these workers are unsuitable for derivation of an RfD.

The toxicity database for oral exposure in laboratory animals includes three chronic duration studies (Chu et al., 1981a; Larson et al., 1979a; NCI, 1976a) and several reproductive and developmental toxicity studies (see Section 4.3 and Table 4-18).

Chu et al. (1981a) fed rats (10/group) chlordecone at 0.07 mg/kg-day for 21 months. The authors reported an increase in liver lesions (described as pericentral cytoplasmic vacuolation with mild anisokaryosis) compared to the control group (5/6 compared to 3/7). Chu et al. (1981a) also reported an increase in thyroid lesions (described as mild degenerative and proliferative changes in the epithelium). However, because of small study size and high incidence of effects in the controls, these increases were not statistically significant (Chu et al., 1981a). Thus, due to limited study size and high incidence of effects in the control group, this study was not selected as the principal study.

NCI (1976a, b) conducted a 20-month feeding study in B6C3F₁ mice and Osborne-Mendel rats. Though treatment groups consisted of 50/sex/group for both rats and mice, only 10 (19 for male mice) matched controls/sex were used. Pooled control groups (from the same laboratory with birth dates within 3–4 months of the treatment groups) contained about 100/sex/group. During the course of the study, toxicity and mortality in the high-dose groups prompted the investigators to reduce dietary chlordecone concentrations one-half to one-sixth of the previous levels. The resulting time-weighted-average dietary concentrations reported by the study authors were 0, 8, or 24 ppm (0, 0.6, or 1.7 mg/kg-day) for male rats and 0, 18, or 26 ppm (0, 1.4, or 2.0 mg/kg-day) for female rats. In mice, time-weighted-average dietary concentrations were 0, 20, or 23 ppm (0, 3.4, or 3.9 mg/kg-day) for male mice and 0, 20, or 40 ppm (0, 3.5, or 7.0 mg/kg-day) for female mice. Noncancer effects reported in response to chlordecone treatment included tremors, dermatologic changes, and liver lesions. The observed liver lesions were characterized as extensive hyperplasia and atypia in both male and female mice in both dose groups. However, due to the lack of incidence data or statistical testing of non-cancer effects, this study was not selected as the principal study.

Larson et al. (1979a) fed groups of Wistar rats (40/sex/group) diets estimated (based on graphically depicted food consumption and body weight data) to result in dose levels of 0, 0.06, 0.3, 0.5, 1.6, 3.9, or 7.0 mg/kg-day for up to 2 years. All rats in the highest two dose groups died within the first 6 months. Though the two highest dose groups were uninformative because of high mortality, four acceptable low-dose exposure groups exist. However due to serial sacrifices and early mortality in several dose groups, effective numbers of animals available for histological examination at the conclusion of the study were greatly reduced with only four animals/sex available in the 1.6 mg/kg-day dose group. The most sensitive effects observed in this study include kidney lesions in females, testicular atrophy in males, and liver lesions in both sexes. The authors reported increased incidence of liver lesions and an increase in relative liver weights in female rats at 0.5 mg/kg-day and male rats at 1.6 mg/kg-day. The liver lesions

observed were characterized primarily as fatty changes and hyperplasia. Testicular atrophy was observed in male rats treated with chlordecone at dose levels of ≥ 1.6 mg/kg-day.

In addition to liver lesions and testicular effects, Larson et al. (1979a) also observed a significant, dose-related increase in the incidence and severity of renal lesions in female Wistar rats in the 0.3, 0.5, and 1.6 mg/kg-day dose groups. The background incidence of renal lesions in male rats was high (55% as compared to 12% in female rats) and, as such, renal effects in male animals did not achieve statistical significance. An increase in proteinuria, a clinical sign of glomerular damage, was observed in female rats, starting at 0.3 mg/kg-day, though data from individual animals were not reported, precluding statistical analysis for this endpoint. Larson et al. (1979a) identified a LOAEL of 0.3 mg/kg-day for proteinuria and increased incidence of glomerulosclerosis in female Wistar rats with a corresponding NOAEL of 0.06 mg/kg-day.

A supporting study by Sobel et al. (2005) found that chlordecone, at doses estimated to be ≥ 0.2 mg/kg-day, increased the severity and decreased the latency of glomerular disease in subcutaneously treated mice of a strain known to be susceptible to autoimmunity mediated glomerulonephritis, (NZB \times NZW) F_1 . Female ovariectomized mice were exposed subcutaneously to sustained-release pellets containing 0.01, 0.1, 0.5, or 1.0 mg chlordecone for up to 30 weeks. Mice treated with 0.5 mg chlordecone pellets (calculated by the authors as an average exposure level of 0.20 mg/kg-day) developed renal impairment (proteinuria and glomerulonephritis) significantly earlier than did ovariectomized controls ($p < 0.05$). Renal sections from the chlordecone-treated mice demonstrated severe proliferative glomerulonephritis with the deposition of immune complexes. A follow-up study by the same group (Sobel et al., 2006), utilizing the same doses and protocol, found that chlordecone treatment of BALB/c mice (a strain not prone to autoimmune disease or glomerular lesions) for up to a year did not produce elevated autoantibody titers or renal disease. Due to the use of s.c. dosing, these studies are considered supportive of the kidney effects, but are not appropriate for the derivation of an oral RfD.

A short-term study in rats provides some additional support for the use of kidney and liver effects as critical effects with chlordecone exposure as observed in the chronic study by Larson et al. (1979a). Chetty et al. (1993c) found significantly elevated serum indicators of kidney (specifically glomerular) and liver damage in male Sprague-Dawley rats (6/group) treated for 15 days with 0, 1, 10, 50, or 100 ppm chlordecone (0.1, 1.0, 4.9, and 9.7 mg/kg-day) in the diet. After 15 days of chlordecone exposure, serum levels of total protein, urea nitrogen, uric acid, creatinine, glutamic oxaloacetic transaminase, glutamic pyruvic transaminase (GPT), ALP, and creatine kinase were measured. GPT was elevated at doses starting at 1.0 mg/kg-day, additionally all other serum enzymes tested were statistically significantly elevated at the highest dose tested (9.7 mg/kg-day). The alterations of serum enzyme levels of liver enzymes suggest chlordecone-induced liver damage. Urea nitrogen was statistically significantly elevated over controls at doses ≥ 4.9 mg/kg-day. At 9.7 mg/kg-day, urea nitrogen, uric acid and creatinine were

statistically significantly elevated. The increased concentrations of urea nitrogen and creatinine in the serum of chlordecone-treated animals suggest kidney dysfunction, likely glomerular in nature (Hart and Kinter, 2005).

Renal effects with chlordecone exposure were also reported in other studies. NCI (1976b) reported chronic kidney inflammation in male (at 0.6 mg/kg-day) and female Osborne-Mendel rats (at 2.0 mg/kg-day). Chu et al. (1980) reported that 28 days of dietary exposure to chlordecone (at 0.07 mg/kg-day) produced eosinophilic inclusions in proximal tubules in 2/10 male Sprague-Dawley rats. A 32-month oral exposure study in beagles (Larson et al., 1979a) reported increased relative kidney weights in the 0.5 mg/kg-day chlordecone exposure group, though renal histology findings were negative. Furthermore, a 3-month oral study observed increased relative kidney weight in female rats exposed to 1.6–1.7 mg/kg-day, though no histological findings were noted (Cannon and Kimbrough, 1979).

Support in the chlordecone database exists for a variety of reproductive effects with chlordecone exposure. Larson et al. (1979a) observed testicular atrophy in male rats treated with chlordecone for 13 weeks at dose levels of ≥ 1.6 mg/kg-day. The incidence of testicular atrophy at 13 weeks was reported as 1/10, 0/5, 1/5, 4/5, 4/5, and 5/5 at 0, 0.3, 0.5, 1.6, 3.9, and 7.0 mg/kg-day. Testicular effects were not noted for the longer exposure durations (1–2 years) in the same study. Other animal studies have shown male reproductive effects, such as decreased sperm viability, motility, and concentration, following exposure to chlordecone (U.S. EPA, 1986c; Linder et al., 1983). U.S. EPA (1986c) reported decreased sperm concentration in male rats treated orally for 10 days with 0.625 mg/kg-day chlordecone. Linder et al. (1983) saw sperm effects (decreased viability, motility, and concentration) in rats at 0.83 and 1.67 mg/kg-day (90 days of treatment); however, the authors did not see any treatment-related histological lesions or an effect on reproductive performance (number of pregnant females, number of live litters, average live litter size, number of implants, percentage of resorptions, and fetal weight) when treated males were mated to untreated females. This study and a study by Cannon and Kimbrough (1979) indicate that decreased reproductive success in experimental animals may not be solely attributable to male reproductive effects. Cannon and Kimbrough (1979) reported that treated female rats (1.6–1.7 mg/kg-day for 3 months) mated to control rats failed to produce litters, whereas treated males (1.2–1.6 mg/kg-day for 3 months) mated to control females had reproductive success similar to controls. Good et al. (1965) reported in a continuous breeding study that male and female mice treated with 0.94 mg/kg-day for 1 month prior to mating and 5 months during mating had impaired reproductive success; reduced production of litters was seen in treated mice and the mated offspring of treated mice. However, the general confidence in this study is limited by incomplete reporting of the variance of reproductive parameters and decreased fertility of the control mice one-generation apart. Another reproductive study (Huber et al., 1965) treated outbred mice in the diet for 1 month prior to mating and 3 months during the mating period with doses of chlordecone starting at 1.9 mg/kg-day and did not see a depression

of reproductive parameters until 5.6 mg/kg-day (Huber et al., 1965). Additional studies have reported reproductive toxicity, but at higher doses (Swartz and Mall, 1989; Swartz et al., 1988; Gellert and Wilson, 1979).

In addition to the reproductive toxicity studies described above, a neurodevelopmental study involved the treatment of female rats 2 months prior to mating, and throughout gestation and lactation, and subjected offspring to neurobehavioral testing at postnatal day (PND) 30 and 100 (Squibb and Tilson 1982). The only neurobehavioral endpoint that was detected was a significant increase in the time required to reorient to a vertical position in an assay for negative geotaxis in male offspring of dams exposed to 0.6 mg/kg-day at 100 days of age. The effect was not seen at 30 days in males and was not seen at either time point in female offspring. Motor activity induced by a dopamine receptor agonist was significantly increased in male offspring at 114 days of age in the high dose group 30 minutes after dosing and both dose groups 60 minutes after dosing. Spontaneous motor activity of treated animals in the absence of pharmacological challenges was not different than controls. In the absence of additional effects suggesting a neurological or behavioral response, the biological significance of the alteration of dopaminergic function in chlordecone-exposed animals following pharmacological challenge is uncertain. Body weights of offspring recorded at day 100 were statistically significantly decreased 27% in females at both 0.1 and 0.6 mg/kg-day and 19% in males at 0.6 mg/kg-day. Recorded body weights at all other time points (PND 1, 7, 14, and 30) were no different from controls. No dose-response relationship was demonstrated in this study for decreased pup body weight in females. A LOAEL was determined based on decreased body weight of female offspring at 100 days following a dietary maternal dose of 0.1 mg/kg-day chlordecone.

The latency of the decreased body weight in adult offspring makes this finding difficult to interpret. Body weight decreases can be indicative of toxicity. However, Squibb and Tilson (1982) reported no visible signs of toxicity in any of the treatment groups, nor were any neurobehavioral effects detected in the female dose groups with depressed weight. Limited data on body weight observations are available from additional developmental studies. Fetal body weight decreases in rats have been observed at higher doses of chlordecone. Chernoff and Rogers (1976) treated pregnant rats and mice on GDs 7–16 with doses of chlordecone ranging from 2–12 mg/kg-day. A LOAEL for decreased fetal body weight in rats was determined as 6 mg/kg-day (with a NOAEL of 2 mg/kg-day), whereas no effect on fetal weight was determined for mice. Another developmental study provided body weight observations of adult offspring of rat dams exposed on GDs 14–20 with 15 mg/kg-day chlordecone (Gellert and Wilson 1979). Body weights of offspring at 6 months of age were not statistically different from controls, with body weight increased 8% in females and decreased 8% in males. However female offspring were not without residual reproductive effects (PVE, anovulation and altered levels of serum estradiol). It should be noted that the dosing period of the Squibb and Tilson study was longer than both the Chernoff and Rogers (1976) and Gellert and Wilson (1979) developmental studies

with exposure including mating, gestation, and lactation. Thus the full gestational exposure and postnatal lactation exposure may have resulted in increased sensitivity of the offspring. It is possible that developmental exposure to chlordecone resulted in a subtle alteration of the endocrine system manifesting in latent decreased growth of adult animals, perhaps linked to the hormonal activity of chlordecone. MOA data to support this hypothesis, however, are not available. In consideration of the uncertainties regarding the finding of decreased body weight in adult female offspring gestationally and lactationally exposed to chlordecone, including the latency, isolation, and lack of dose response, this effect was not considered the most appropriate effect on which to base the derivation of the RfD.

In consideration of the available studies reporting effects of chronic and subchronic chlordecone exposure in humans and animals, Larson et al. (1979a) was chosen as the principal study. This study was designed with several acceptable dose groups and adequate numbers of animals (though numbers of animals in high dose groups were greatly reduced following serial sacrifices and early mortality). Results were sufficiently reported for most endpoints. Sensitive endpoints identified in this study include glomerulosclerosis, liver lesions, and testicular atrophy. Though testicular atrophy was observed at 13 weeks, the only lesions observed chronically that were reported to be treatment related were in the liver and kidney. This observation coupled with the lack of support for testicular lesions in other studies in rats of similar dose and duration (Linder et al., 1983; Cannon and Kimbrough, 1979) decreases confidence in this endpoint. Additionally, the liver lesions observed in the principal study (characterized as fatty changes and hyperplasia) occurred at higher doses as compared with the observed kidney lesions. After consideration of all endpoints, the increased incidence of glomerulosclerosis in female rats was determined to be the most sensitive and biologically significant effect detected in this study. Furthermore, the chlordecone database contains additional support for the specific endpoint of glomerular damage (Sobel et al., 2006, 2005; Chetty et al., 1993c) and general support for the kidney as a target organ as determined by increased kidney weights seen in other studies (Cannon and Kimbrough, 1979; NCI, 1976a).

Glomerulosclerosis is believed to be an irreversible effect that can result in renal impairment (Medical College of Wisconsin, 1999). The mechanism by which chlordecone causes kidney lesions is not known; however, there is no indication that kidney lesions would not occur in humans chronically exposed to chlordecone. Though clinical indications of kidney dysfunction were not detected in workers occupationally exposed to chlordecone, this may be because the relatively short average exposure duration of workers (5–6 months) was not sufficient for the development of detectable kidney impairment. Therefore, for the above reasons, Larson et al. (1979a) was chosen as the principal study and renal lesions as the critical effect.

Several studies described above demonstrate reproductive effects following chlordecone exposure at levels slightly higher than the level reported to cause renal lesions in chronically

treated rats (Linder et al., 1983; Cannon and Kimbrough, 1979; Larson et al., 1979a; Good et al., 1965; Huber et al., 1965). Therefore, reproductive effects were not selected as the critical effect of chlordecone exposure. However, potential points of departure (PODs) for reproductive endpoints from Linder et al. (1983), Squibb and Tilson (1982), Larson et al. (1979a), and Good et al. (1965) are presented for comparison (see Section 5.1.2 and Appendix B).

5.1.2. Methods of Analysis

All available models in the U.S. EPA Benchmark Dose Software (BMDS) version 1.3.2 were fit to quantal incidence data for histopathologic renal lesions in female Wistar rats from a 2-year dietary study (Larson et al., 1979a). The data modeled are shown below in Table 5-1.

Table 5-1. Incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in male or female Wistar rats following administration of chlordecone in the diet for 1–2 years

| Gender | Dose (mg/kg-day) | | | | |
|---------------------|------------------|------------|-------------------------|-------------------------|------------------------|
| | 0 | 0.06 | 0.3 | 0.5 | 1.6 |
| Male | 12/22 (55%) | 3/11 (27%) | 4/6 (67%) | 6/9 (67%) | 3/4 (75%) |
| Female ^a | 4/34 (12%) | 2/13 (15%) | 8/17 (47%) ^b | 8/12 (67%) ^b | 3/4 (75%) ^b |

^aStatistically significant trend for increased incidence by Cochran-Armitage test ($p < 0.01$).

^bStatistically significantly different from controls according to Fisher's exact test ($p < 0.05$) performed for this review.

Source: Larson et al. (1979a).

Biological and statistical considerations were taken into account in the selection of a benchmark response (BMR) level for this data set. Statistically, a 10% level of response is intended to represent a response level near the lower range of detectable observations in typical studies conducted with 50 animals per dose group (U.S. EPA, 2000c). The data set for the critical effect from Larson et al. (1979a) relies on notably smaller groups of animals (4–22 animals/group); therefore, use of a BMR below 10% would result in a POD further outside of the observable range and would involve greater uncertainty. Biologically speaking, a BMR of a 10% increase in glomerulosclerosis was selected under an assumption that it represents a minimal biologically significant change (U.S. EPA, 2000c). Therefore, for this dataset, a response level of 10% was used. The results of benchmark dose (BMD) modeling of the data are discussed below.

Statistical analysis of the incidence of glomerulosclerosis (grades 1, 2, or 3 combined) in each dose group by sex revealed that the incidence of glomerulosclerosis in female rats exhibited a significant dose response trend (according to the Cochran-Armitage test). Therefore, the models within the BMDS were fit to the incidence data for renal lesions in female rats in the 0, 0.06, 0.3, 0.5, and 1.6 mg/kg-day dose groups to derive BMD_{10S} and 95% lower confidence limit

on the BMD₁₀ (BMDL_{10S}). It should be noted that all animals from the two highest dose groups (3.9 and 7.0 mg/kg-day) died within the first 6 months of the study, and thus data from these animals were not available for use in the dose-response assessment.

As shown in Appendix B, most of the models in BMDS (version 1.3.2.) provided adequate fits to the incidence data for histopathologic renal lesions (glomerulosclerosis) in female rats from the Larson et al. (1979a) study (Table 5-1), as assessed by a chi-square goodness-of-fit test (i.e., models with $p < 0.1$ failed to meet the goodness-of-fit criterion) and the Akaike's Information Criterion (AIC) value (i.e., a measure of the deviance of the model fit that allows for comparison across models for a particular endpoint). BMDL₁₀ estimates from these models were within a factor of three of each other suggesting no appreciable model dependence. The log-probit model provided the best fit to the female rat data as assessed by the AIC. Thus, the log-probit model was selected to estimate the BMD for glomerulosclerosis data in female rats from Larson et al. (1979a). The BMD₁₀ associated with a 10% extra risk for glomerulosclerosis in female rats was 0.12 mg/kg-day, and the BMDL₁₀ was 0.08 mg/kg-day. The incidence of liver lesions (fatty change and hyperplasia) in rats was also modeled yielding BMD₁₀ and BMDL₁₀ estimates of 0.23 and 0.14 mg/kg-day, respectively.

Reproductive effects observed following oral exposure to chlordecone were also evaluated as potential PODs. Reproductive endpoints, such as decreased sperm concentration (Linder et al., 1983) and testicular atrophy (Larson et al., 1979a) along with functional reproductive outcomes, such as decreases in first and second litters (Good et al., 1965; Huber et al., 1965), were investigated. The incidence of testicular atrophy in male Wistar rats, following 3 months of dietary chlordecone exposure (Larson et al., 1979a) was also modeled. The BMD₁₀ associated with a 10% extra risk for testicular atrophy in rats was 0.21 mg/kg-day, and its BMDL₁₀ was 0.12 mg/kg-day. BMD modeling of the decreased sperm concentration associated with one standard deviation from the control mean observed in Linder et al. (1983) identified a BMD_{1SD} of 1.36 mg/kg-day and a BMDL_{1SD} of 0.86 mg/kg-day. Modeling results for these endpoints are included as part of Appendix B.

The continuous reproductive endpoints reported (percent of pairs producing first and second litters, pair days/litter) by Good et al. (1965) were averages and did not include any measure of the variability, such as standard deviations, thus it was determined that these data were not amenable to BMD modeling. A freestanding LOAEL of 0.94 mg/kg-day was identified for the reduced production of second litters in chlordecone treated BALB/c mice and reduced reproduction in offspring of treated mice (reduced production of first litters) (Good et al., 1965). Due to identical response levels in female offspring at both doses (0.1 and 0.6 mg/kg-day), the data reported by Squibb and Tilson (1982) were not amenable to BMD modeling. However, a freestanding LOAEL of 0.1 mg/kg-day was identified based on decreased body weight in female offspring of treated rats (Squibb and Tilson, 1982).

5.1.3. RfD Derivation—Including Application of Uncertainty Factors (UFs)

Of the endpoints shown in Table 4-18, the increased incidence of histopathological renal lesions (glomerulosclerosis) among female Wistar rats receiving chlordecone in the diet continuously for 2 years (Larson et al., 1979a) is the most sensitive endpoint. BMD modeling revealed that the BMDL₁₀ associated with this effect is 0.08 mg/kg-day. The BMDL₁₀ provides the POD for the RfD.

A total UF of 300 was applied to the POD of 0.08 mg/kg-day: 10 for interspecies extrapolation from animals to humans (UF_A); 10 for human intraspecies variability (UF_H); and 3 to account for database deficiencies (UF_D).

An UF of 10 was used to account for uncertainties in extrapolating from laboratory rats to humans. Aside from a difference in metabolism (humans produce chlordecone alcohol, whereas rats do not), the available data do not suggest differential toxicity of these forms, nor do the toxicity data from various animal species provide evidence that rats or any other species are more sensitive to chlordecone than humans. Consequently, the default UF of 10 for extrapolating from laboratory animals to humans was applied.

An UF of 10 was used to account for variation in susceptibility among members of the human population (i.e., interindividual variability). Insufficient information is available to predict potential variability in human susceptibility.

An UF of 3 was applied to account for deficiencies in the chlordecone toxicity database. The database includes limited human data from observational studies of occupationally exposed workers. The database also includes several studies in laboratory animals, including chronic and subchronic dietary exposure studies and several subchronic reproductive and developmental studies, as well as one study specifically assessing developmental neurotoxicity. The chlordecone database does not have a standard multigenerational reproductive study, but includes approximately 10 oral repeat-exposure studies assessing reproductive and developmental toxicity, including several single-generation reproductive toxicity studies and three developmental studies in rats and mice (Linder et al., 1983; Squibb and Tilson, 1982; Cannon and Kimbrough, 1979; Chernoff and Rogers, 1976; Good et al., 1965; Huber et al., 1965). Several of these reproductive studies have indicated decreased reproductive success in chlordecone-treated animals at doses higher than those associated with kidney lesions (Cannon and Kimbrough, 1979; Good et al., 1965; Huber et al., 1965). The database also includes two nonstandard multigenerational studies that evaluated reproductive success of chlordecone-treated animals (Gellert and Wilson, 1979; Good et al., 1965). Due to limited scope and design, these studies are not considered adequate for the assessment of potential multigenerational reproductive toxicity. Therefore, in consideration of the entire database for chlordecone, a database UF of 3 is considered appropriate to account for the lack of a two-generational reproductive study.

Because the POD was selected from a dose associated with an endpoint identified by a chronic dietary study (Larson et al., 1979a), no uncertainty factor is needed for exposure duration (subchronic to chronic). An UF for LOAEL-to-NOAEL extrapolation was not used because the current approach is to address this factor as one of the considerations in selecting a BMR for BMD modeling. In this case, a BMR of a 10% increase in glomerulosclerosis was selected under an assumption that it represents a minimal, biologically significant change.

The oral RfD for chlordecone was calculated as follows:

$$\begin{aligned}\text{RfD} &= \text{BMDL}_{10} \div \text{UF} \\ &= 0.08 \text{ mg/kg-day} \div 300 \\ &= 0.0003 \text{ or } 3 \times 10^{-4} \text{ mg/kg-day}\end{aligned}$$

5.1.4. Reference value (RfV) Comparison Information

Kidney (glomerular) lesions, liver lesions, testicular atrophy, and decreased fertility are observed low-level effects, following subchronic or chronic oral exposure to chlordecone (Larson et al., 1979a; Good et al., 1965). Table 5-2 provides a tabular summary of alternate PODs and resulting potential RfVs for these endpoints. Additionally, Figure 5-1 provides a graphical representation of this information. This figure should be interpreted with caution since the PODs across studies are not necessarily comparable, nor is the confidence the same in the data sets from which the PODs were derived. The PODs presented in this figure are based on either a BMDL_{10} (for kidney, testicular, or liver lesions), a BMDL_{1SD} (for decreased sperm concentration), or a LOAEL (for decreased production of litters or decreased female offspring weight). Some indication of the confidence associated with the resulting potential RfVs are reflected in the magnitude of the total UF applied to the POD (i.e., the size of the bar); however, the text of Sections 5.1.1 and 5.1.2 should be consulted for a more complete understanding of the issues associated with each dataset and the rationale for the selection of the principal study and the critical effect used to derive the RfD. As discussed in Section 5.1.1, among the studies considered, the chronic study by Larson et al. (1979a) provided the data set most appropriate for the derivation of the RfD.

Table 5-2. Possible PODs with applied uncertainty factors and resulting Potential RfVs

| Effect | POD | Species | Uncertainty factors ^a | | | | | Potential RfV | |
|---------------------------------|-------------------|---------|----------------------------------|----|----|----|---|---------------|--------------------|
| | | | Total | A | H | L | S | | D |
| Kidney lesions | 0.08 ^b | Rat | 300 | 10 | 10 | | | 3 | 3×10^{-4} |
| Decreased offspring weight | 0.1 ^c | Rat | 3,000 | 10 | 10 | 10 | | 3 | 3×10^{-5} |
| Testicular atrophy | 0.12 ^b | Rat | 300 | 10 | 10 | | | 3 | 4×10^{-4} |
| Liver lesions | 0.14 ^b | Rat | 300 | 10 | 10 | | | 3 | 5×10^{-4} |
| Decreased sperm count | 0.86 ^d | Rat | 300 | 10 | 10 | | | 3 | 3×10^{-3} |
| Decreased production of litters | 0.94 ^c | Mouse | 3,000 | 10 | 10 | 10 | | 3 | 3×10^{-4} |

^aUncertainty factors: A = animal to human (interspecies); H = interindividual (intraspecies); L = LOAEL to NOAEL; S = subchronic-to-chronic duration; D = database deficiency.

^bPOD based on BMDL determined through BMD modeling of a 10% response. Source: Larson et al. (1979a).

^cPOD based on freestanding LOAEL. Source: Squibb and Tilson (1982).

^dPOD based on BMDL determined through BMD modeling of a 1 standard deviation change. Source: Linder et al. (1983).

^ePOD based on a freestanding LOAEL. Source: Good et al. (1965).

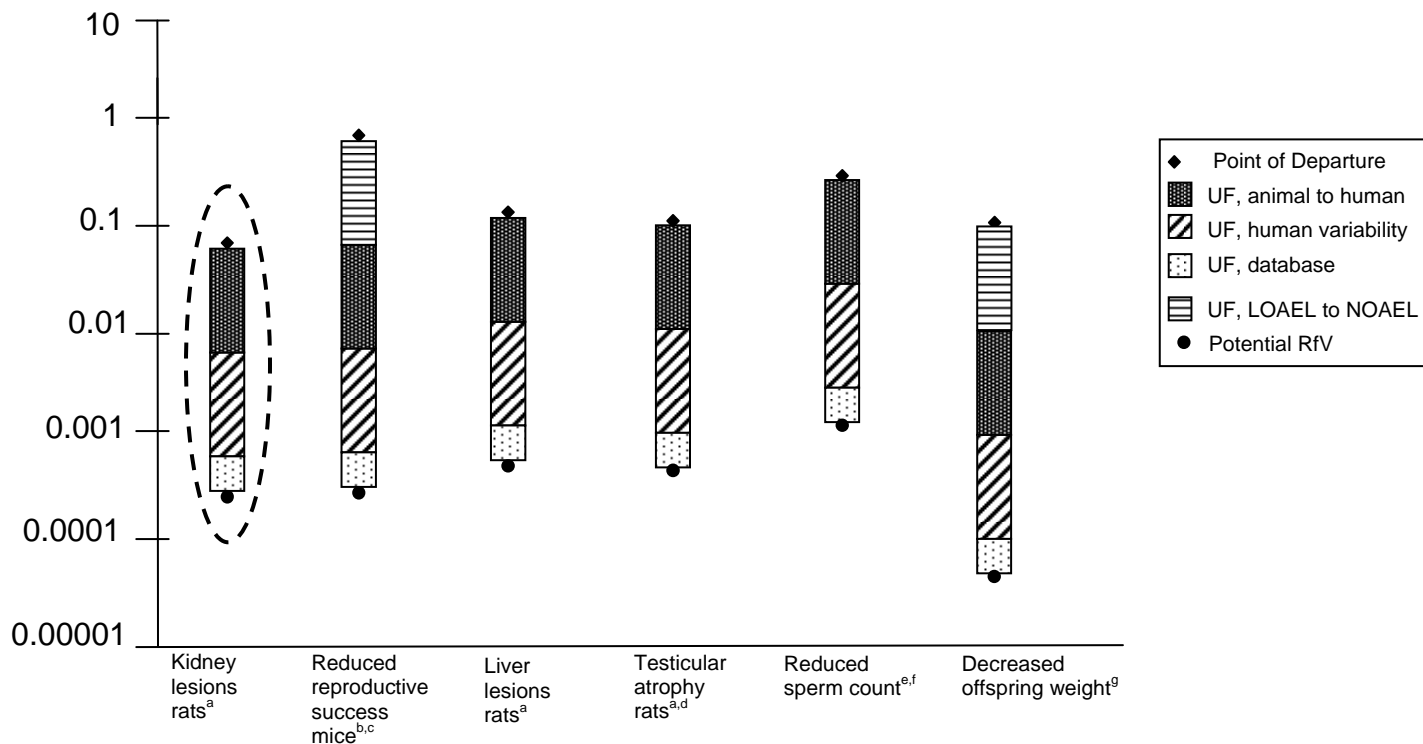


Figure 5-1. Potential RfV comparison array for alternate points of departure.

^aLarson et al. (1979a).

^bGood et al. (1965).

^cPOD based on a freestanding LOAEL for a 65% decrease in second-generation animals producing litters.

^dSubchronic endpoint (13 weeks).

^eLinder et al. (1983).

^fBMDL_{1SD} used as the POD.

^gSquibb and Tilson (1982).

The PODs presented for kidney, liver, and testicular lesions were derived through BMD modeling of the dichotomous data using a 10% response level. The POD presented for reduced sperm count was derived through BMD modeling of continuous data using a response level of one standard deviation from the control mean. BMD modeling outputs for these endpoints are included in Appendix B. The PODs based on BMD methods have an inherent advantage over the use of a NOAEL or LOAEL by making greater use of all the dose-response data from a given data set. The PODs for reduced reproductive success in mice and decreased offspring weight in rats were based on freestanding LOAELs.

The POD for testicular atrophy was derived from a 3-month exposure duration (within the chronic study by Larson et al. [1979a]); however, testicular effects were not noted for the longer exposure durations (1–2 years) in the same study, nor were testicular lesions detected in other studies in rats treated with similar doses for the same duration (Linder et al., 1983; Cannon and Kimbrough, 1979). Because testicular atrophy was not detected at the chronic timepoint in the same study, an uncertainty factor to account for the use of a subchronic duration was not applied to the potential POD for this endpoint.

5.1.5. Previous RfD Assessment

An oral assessment for chlordecone was not previously available on IRIS.

5.2. INHALATION REFERENCE CONCENTRATION (RfC)

Although adverse health effects from an occupational exposure incident may have resulted from inhalation exposure (in combination with oral and dermal exposures), the data do not identify exposure concentrations at which the effects occur (Taylor, 1985, 1982; Guzelian, 1982a; Guzelian et al., 1980; Sanborn et al., 1979; Cannon et al., 1978; Martinez et al., 1978; Taylor et al., 1978). Consequently, the human data cannot be used to define an exposure-response relationship for inhalation exposure to chlordecone. No studies on the toxicity of chlordecone following inhalation exposure in laboratory animals were located. This lack of data precludes the derivation of an RfC.

Consideration was given to route-to-route extrapolation to derive inhalation doses from existing oral dose-response data for development of an RfC. Route-to-route extrapolation from the oral database, however, is precluded by deficiencies in the database. The available rat PBTK models for chlordecone do not include the inhalation route of exposure (see Section 3.5), and human PBTK models with both oral and inhalation portals of entry have not yet been developed. In the absence of PBTK models that include oral and inhalation routes of exposure, and lacking inhalation absorption efficiency data in humans and rats, a route-to-route extrapolation from oral to inhalation for chlordecone would be highly uncertain. As discussed in Chapter 2, only very small amounts of chlordecone will evaporate from soil or water surfaces, and any chlordecone in the air is likely to be removed by deposition of particles.

5.3. CANCER ASSESSMENT

There are no human studies that assess carcinogenic potential of chlordecone. An 80 week dietary study in male and female Osborne-Mendel rats and B6C3F1 mice provides evidence of chlordecone-induced liver tumors in both sexes of two species. The mode of action of the liver tumors observed is unknown, thus, in the absence of this information, the tumors are considered relevant to the assessment of the carcinogenic potential of chlordecone in humans. Utilizing the EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), chlordecone is "likely to be carcinogenic to humans".

5.3.1. Choice of Study/Data with Rationale and Justification

Of the available oral chronic toxicity studies in animals (Chu et al., 1981a; Larson et al., 1979a; NCI, 1976a), only the cancer bioassay of chlordecone by NCI (1976a) found evidence of carcinogenicity. The remaining chronic studies of chlordecone (Chu et al., 1981a; Larson et al., 1976a) lacked adequate power to detect carcinogenicity. Chu et al. (1981a) included only one dose group of 10 animals/sex and did not use an adequately high dose (0.07 mg/kg-day). The study by Larson et al. (1979a) also was limited in power. Specifically, only four animals/sex were examined in the highest dose group (1.6 mg/kg-day) at the termination of the study. The NCI (1976a) bioassay involved the administration of chlordecone in the diet at two doses in both sexes of two rodent species (rat and mouse). This study included 50 animals per sex per dose group, though the number of matched controls was less than optimal (n = 10–20). However, these concurrent controls were compensated for by additional control groups from the same laboratory with birthdates within 3–4 months of the exposed groups (referred to as "pooled controls"). Histopathologic examination of a wide variety of tissues and organs was performed. High toxicity and early mortality in some treated animals at initial doses resulted in the authors lowering the doses. Tumor incidences in the liver were elevated with increasing exposure levels across all sex/species combinations compared to pooled controls. A statistically significant dose-response trend was observed in both sexes of rats and in male mice (Cochran-Armitage test). The incidences of hepatocellular carcinoma in female mice were roughly the same in the low and high dose group (52 and 47%) and were statistically significant compared to control incidence (0%) in pairwise comparisons (Fisher's exact test). Decreased survival of high dose animals reduced the sensitivity of the study in some dose groups but did not prevent the detection of statistically significantly elevated incidences of hepatocellular carcinoma in both sexes of rats and mice. Other than those in the liver, no other tumors were statistically significantly elevated above controls in the chlordecone-treated groups of either rats or mice.

The mode of carcinogenic action of chlordecone in the livers of rats and mice is unknown. Most genotoxicity tests using chlordecone are negative. For liver tumors in rats and mice, some data suggest that chlordecone may induce cell proliferation and lead to a promotion

in the growth of pre-initiated cells. However, key precursor events linked to observed cell proliferation have not been identified, and thus the mode of action for liver tumors has not been established. Because the mode of action is unknown, the liver tumors observed in the NCI cancer bioassay in rats and mice are considered relevant to the assessment of the carcinogenic potential of chlordecone in humans.

This study exhibits several methodological issues which limit somewhat the level of confidence in the quantification of cancer risk but do not negate the findings of statistically elevated liver tumors in two sexes of two species. The limitations include inconsistent dose levels, use of only two dose groups, high early toxicity and reduced survival, and low numbers of matched controls (10–20/sex/group). Some mitigation of these limitations was accomplished by the inclusion of pooled controls and the use of time-to-tumor modeling to account for high toxicity and resulting early deaths and use of a lifetime average dose in the modeling of the data. As in all risk assessments, uncertainties exist in this analysis. Section 5.3.5. reviews the key uncertainties in the use of this study to estimate potential risks to human populations from exposure to chlordecone.

5.3.2. Dose-Response Data

In the NCI (1976a) study, groups of 50 male and female Osborne-Mendel rats and B6C3F₁ mice were administered chlordecone in the diet for 80 weeks. The initial dietary concentrations were reduced at least once in each dose group during the course of the study because they were not well tolerated. Average doses reported by the NCI study authors were: 8 and 24 ppm (male rats), 18 and 26 ppm (females rats), 20 and 23 ppm (male mice), and 20 and 40 ppm (female mice). Dosing was concluded after 80 weeks and all surviving rats were sacrificed at 112 weeks, while all surviving mice were sacrificed at 90 weeks. Statistically significant increased incidences of hepatocellular carcinomas were observed in both sexes of rats and mice. These tumors appeared earlier with increasing exposure levels in rats and mice, and showed statistically significant increasing trends with increasing exposure levels in both sexes of rats. These data are summarized in Table 5-3 (for male and female rats) and Table 5-4 (for male and female mice).

Table 5-3. Tumor incidence and time to first tumor for hepatocellular carcinomas observed in Osborne-Mendel rats following administration of chlordecone in the diet for 80 weeks

| Gender | Parameter | Exposure group | | | |
|--------|-----------------------------|-----------------|----------------|-----------|-----------------------------|
| | | Matched control | Pooled control | Low dose | High dose |
| Male | Tumor incidence | 0/10 | 0/105 | 1/50 (2%) | 3/44 (7%) ^{a, b} |
| | Time to first tumor (weeks) | NA | NA | 112 | 108 |
| Female | Tumor incidence | 0/10 | 0/100 | 1/49 (2%) | 10/45 (22%) ^{a, b} |
| | Time to first tumor (weeks) | NA | NA | 87 | 83 |

^aStatistically significant dose response trend ($p < 0.05$) by Cochran-Armitage trend test.

^bStatistically significant increase in incidence, as compared with pooled controls, using one-tailed ($p < 0.05$) Fisher's exact test for 2×2 contingency table.

Source: NCI (1976a).

Table 5-4. Tumor incidence and time to first tumor for hepatocellular carcinomas observed in B6C3F₁ mice following administration of chlordecone in the diet for 80 weeks

| Gender | Parameter | Exposure group | | | |
|--------|-----------------------------|-----------------|----------------|--------------------------|-----------------------------|
| | | Matched control | Pooled control | Low dose | High dose |
| Male | Tumor incidence | 6/19 (31%) | 8/49 (16%) | 39/48 (81%) ^b | 43/49 (88%) ^{a, b} |
| | Time to first tumor (weeks) | 87 | 87 | 70 | 62 |
| Female | Tumor incidence | 0/10 | 0/40 | 26/50 (52%) ^b | 23/49 (47%) ^{a, b} |
| | Time to first tumor (weeks) | NA | NA | 87 | 76 |

^aStatistically significant dose response trend ($p < 0.05$) by Cochran-Armitage trend test.

^bStatistically significant increase in incidence as compared with matched or pooled controls, using one-tailed ($p < 0.05$) Fisher's exact test for 2×2 contingency table.

Source: NCI (1976a).

5.3.3. Dose Adjustments and Extrapolation Methods

The U.S. EPA *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a) recommend that the method used to characterize and quantify cancer risk from a chemical is determined by what is known about the mode of action of the carcinogen and the shape of the cancer dose-response curve. The linear approach is used as a default option if the mode-of-action of carcinogenicity is not understood (U.S. EPA, 2005a). In the case of chlordecone, the mode of carcinogenic action of chlordecone in the livers of rats and mice is unknown. Therefore, a linear low-dose extrapolation approach was used to estimate human carcinogenic risk associated with chlordecone exposure.

Due to the earlier occurrence of tumors with increasing exposure and the mortality observed (especially in the high-dose groups in the second year of the study), dose-response methodologies which can account for the influence of competing risks and intercurrent mortality on site-specific tumor incidence rates are preferred. The U.S. EPA has generally used a model which incorporates the time at which death-with-tumor occurred as well as the dose; the multistage-Weibull model is multistage in dose and Weibull in time, and has the form:

$$P(d) = 1 - \exp[-(q_0 + q_1d + q_2d^2 + \dots + q_kd^k) \times (t - t_0)^z],$$

where $P(d)$ represents the lifetime risk (probability) of cancer at dose d (i.e., human equivalent exposure in this case); parameters $q_i \geq 0$, for $i = 0, 1, \dots, k$; t is the time at which the tumor was observed; and z is a parameter which characterizes the change in response with age. The parameter t_0 represents the time between when a potentially fatal tumor becomes observable and when it causes death, and is generally set to 0 either when all tumors are considered incidental or because of a lack of data to estimate the time reliably. The dose-response analyses in this assessment were conducted using the computer software program TOX_RISK, Version 5.3 (ICF, Fairfax, VA), which is based on Weibull models drawn from Krewski et al. (1983). Parameters were estimated using the method of maximum likelihood.

Time-to-tumor analysis, as implemented by the TOX_RISK software program, allows the distinction between tumor types that are fatal versus incidental in order to adjust for competing risks. Incidental tumors are those tumors thought not to have caused the death of an animal, while fatal tumors are thought to have resulted in animal death. The NCI (1976a) study did not report individual causes of death, which would be preferable for time-to-tumor analysis. Thus, all liver tumors observed were classified as incidental for the purpose of this analysis, and consequently t_0 was set to zero. The data input into TOX_RISK, as well as the model output, are provided in Appendix C.

Lifetime average dietary concentrations were calculated by TOX-RISK. Dietary concentrations were reported as follows: 5.2 and 15.7 ppm (male rats), 12.2 and 17.9 ppm (female rats), 16.8 and 19.6 ppm (male mice), and 16.8 and 33.7 ppm (female mice).

Average daily doses per unit bodyweight were calculated for this review based on average animal bodyweights and food consumption (US EPA 1988) as 0.36 and 1.1 mg/kg-day (male rats), 0.94 and 1.4 mg/kg-day (female rats), 2.9 and 3.4 mg/kg-day (male mice), and 2.9 and 5.8 mg/kg-day (female mice).

For the incidence of hepatocellular carcinomas, specific *n*-stage Weibull models were selected for each species and sex based on the values of the log-likelihoods according to the strategy used by U.S. EPA (2002). If twice the difference in log-likelihoods was less than a χ^2 with degrees of freedom equal to the difference in the number of stages included in the models being compared, the models were considered comparable, and the most parsimonious model (i.e., the lowest-stage model) was selected. For tumors treated as incidental, plots of model fits compared with Hoel-Walburg estimates of cumulative incidence were also examined for goodness of fit in the lower exposure region of the observed data (Gart et al., 1986). If the model with the additional stage fitted the data in the low-dose region better than the more parsimonious (or lower stage) model, then the model with the additional stage was selected.

Points of departure for estimating low-dose risk were identified at doses at the lower end of the observed incidence data, generally corresponding to 10% extra risk, where extra risk is defined as $[P(d) - P(0)]/[1 - P(0)]$. The lifetime oral cancer slope factor for humans is defined as the slope of the line drawn from the lower 95% bound on the exposure at the POD to 0. This 95% upper confidence limit on the slope represents a plausible upper bound on the true risk.

Adjustments for approximating human equivalent slope factors applicable to continuous exposures over a lifetime were carried out by the TOX_RISK dose-response software program. Consistent with the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), an adjustment for cross-species scaling was applied by the software program to address toxicological equivalence across species after the model-fitting phase. Following U.S. EPA's cross-species scaling methodology, the time-weighted daily average doses were converted to human equivalent doses on the basis of $(\text{body weight})^{3/4}$ (U.S. EPA, 1992). Time-weighted average doses were estimated by TOX-RISK based on the following data inputs for each species and sex:

- Dose level (as ppm in food)
- Days/week of exposure = 7
- Hours/day of exposure = 24
- Duration of exposure (in weeks)
- Adult body weight (in kg)
- Food consumption (in g/day)

Adult body weights and food consumption were taken from U.S. EPA (1988). Dose level and duration of exposure were taken from Table I of the NCI (1976a) final report. Appendix C lists the values employed for these data inputs for each dosing period of the study.

5.3.4. Derivation of the Oral Cancer Slope Factor

The results of applying the multistage-Weibull model implemented in TOX_RISK to the liver tumor incidence data and dosing information for the four species/sex combinations in the NCI (1976a) study are provided in Table 5-5. This table presents the modeling results from the “best-fit” model identified for each species/sex combination using the criteria described above (i.e., change in value of the log-likelihood and visual fit in the low-dose region). An oral slope factor based on the incidence of hepatocellular carcinomas for each species/sex combination was calculated by dividing the BMR (10%) by its corresponding BMDL. For hepatocellular carcinomas in male rats, TOX_RISK failed to estimate multistage-Weibull model coefficients (except for z) and yielded a BMD₁₀ that was “unbounded.” These results suggest a failure of the model-fitting algorithm for these data. Therefore, the slope factor based on this endpoint in male rats was not further considered.

Based on the modeling results summarized in Table 5-5, the recommended oral cancer slope factor for use in estimating human cancer risk from continuous lifetime oral exposure to chlordecone is 9.89, rounded to 10 (mg/kg-day)⁻¹. This slope value was selected primarily because male mice are the most sensitive to tumor induction following exposure to chlordecone. The oral slope factor is derived from the BMDL₁₀, the 95% lower bound on the dose associated with a 10% extra cancer risk of hepatocellular carcinoma in male B6C3F₁ mice, by dividing the BMR (0.10) by the BMDL₁₀, and represents an upper bound, continuous lifetime exposure estimate of cancer potency:

The BMDL₁₀, the lower 95% bound on exposure at 10% extra risk, is 1.01×10^{-2} mg/kg-day and the slope of the linear extrapolation from the BMDL₁₀ to 0 = $0.10/1.01 \times 10^{-2} = \mathbf{10 \text{ per mg/kg-day}}$.

This slope factor should not be used with chlordecone exposures greater than 0.01 mg/kg/day because the observed dose-response relationship does not continue linearly above this dose level. The fitted dose-response model better characterizes what is known about the carcinogenicity of chlordecone above 0.01 mg/kg/day.

Table 5-5. Summary of time-to-tumor dose-response modeling based on the incidence of liver tumors in Osborne-Mendel rats and B6C3F₁ mice

| Tumor Type ^a | Multistage-Weibull model coefficients (MLE) ^b | Human equivalent dose (mg/kg-day) ^c | | Slope factor ^d (mg/kg-day) ⁻¹ |
|------------------------------|--|--|-------------------------|---|
| | | BMD ₁₀ | BMDL ₁₀ | |
| Male rats^e | | | | |
| Female rats | | | | |
| Hepatocellular carcinoma | q ₀ = 0.00 q ₁ = 0.00 q ₂ = 1.24 × 10 ⁻²⁴ z = 10.00 | 2.92 × 10 ⁻¹ | 5.75 × 10 ⁻² | 1.74 |
| Male mice | | | | |
| Hepatocellular carcinoma | q ₀ = 5.16 × 10 ⁻²¹ q ₁ = 3.05 × 10 ⁻²¹ z = 10.00 | 1.50 × 10 ⁻² | 1.01 × 10 ⁻² | 9.89 |
| Female mice | | | | |
| Hepatocellular carcinoma | q ₀ = 0.00 q ₁ = 4.78 × 10 ⁻²² z = 10.00 | 9.53 × 10 ⁻² | 7.24 × 10 ⁻² | 1.38 |

^aAll tumors of the type listed were considered incidental to the death of the animal.

^bMultistage-Weibull model: $P(d) = 1 - \exp[-(q_0 + q_1d + q_2d^2 + \dots + q_kd^k) \times (t - t_0)^z]$, with coefficients estimated by TOX_RISK using methods of maximum likelihood in terms of mg/kg-day as administered in the NCI (1976a) rodent bioassay.

^cPoints of departure adjusted to estimate human equivalent continuous exposure, using BW^{3/4} cross-species scaling.

^dSlope factors estimated by dividing the BMR (10%) by the BMDL.

^eModel fitting failed for this dataset

Source: NCI (1976a).

5.3.5. Uncertainties in Cancer Risk Values

As in most risk assessments, extrapolation of study data to estimate potential risks to human populations from exposure to chlordecone involves some inherent uncertainty. Several types of uncertainty may be considered quantitatively, but other important uncertainties cannot be considered quantitatively. Thus, an overall integrated quantitative uncertainty analysis is not presented. Principal uncertainties are summarized below and in Table 5-6.

Table 5-6. Summary of uncertainty in the chlordecone cancer risk assessment

| Consideration/ approach | Impact on oral slope factor | Decision | Justification |
|--|--|--|---|
| Low-dose extrapolation procedure | Alternatives could ↓ or ↑ slope factor by an unknown extent | Multistage-Weibull model to determine POD, linear low-dose extrapolation from POD | A linear-low-dose extrapolation approach was used to estimate human carcinogenic risk associated with chlordecone exposure. Due to the lack of MOA data to inform the selection of a dose-response model, the linear approach is used in the absence of an alternative. |
| Dose metric | Alternatives could ↑ or ↓ slope factor by an unknown extent | Used administered exposure | Experimental evidence supports a role for limited metabolism in humans, but not in rats or mice. If the target dose in humans is proportional to administered exposure, the slope factor provides an unbiased estimate of risk. |
| Cross-species scaling | Alternatives could ↓ or ↑ slope factor e.g., sixfold ↓ (scaling by BW) or ↑ twofold (scaling by $BW^{2/3}$) | $BW^{3/4}$ (default approach) | There are no data to support alternatives. Because the dose metric was not an AUC, $BW^{3/4}$ scaling was used to calculate equivalent cumulative exposures for estimating equivalent human risks (U.S. EPA, 1992). |
| Statistical uncertainty at POD | ↓ slope factor 1.5-fold if BMD used rather than lower bound on POD | $BMDL_{10}$ (default approach for calculating reasonable upper bound slope factor) | Size of bioassay results in sampling variability; lower bound is 95% confidence interval on administered exposure. |
| Bioassay–exposure issues | Human risk could ↓ or ↑, if continuous lifetime exposure was not estimated correctly | NCI study | Alternative bioassays were inconclusive; exposures in NCI study were not constant throughout the animals' lifetime, nor administered for the typical 104 weeks. |
| Species/gender combination | Human risk could ↓ or ↑, depending on relative sensitivity | Male mouse liver tumors | It was assumed that humans are as sensitive as the most sensitive rodent gender/species tested; true correspondence is unknown. The carcinogenic response occurs across animal species, lending support to its human relevance; liver is a target organ in humans for noncancer toxicity. |
| Human population variability in metabolism and response/sensitive subpopulations | Low-dose risk ↑ to an unknown extent | Considered qualitatively | No data to support range of human variability/sensitivity.. |

Bioassay selection

The study by NCI (1976a, b) was used for development of an oral slope factor. This study was conducted in both sexes in two species which examined a range of toxicological endpoints. The dose selection for this study initially exceeded the maximum tolerated dose, but was subsequently lowered to be better tolerated by the animals. This change in protocol has an unknown impact on the estimated equivalent lifetime exposure.

While 50 test animals were allocated among two dose levels, the number of matched controls was less than optimal ($n = 10\text{--}20$). However, these concurrent controls were compensated for by additional control groups from the same laboratory with birthdates within 3–4 months of the matched control and exposed groups. The increased response in all species/sex combinations but the male rat was statistically significant ($p < 0.05$), even using the small matched control groups, while that in the male rats was statistically significant when compared with the pooled controls. Overall, responses across the four species/sex combinations consistently indicated increased incidences of hepatocellular carcinogenicity. Alternative chronic bioassays lacked sufficient power to detect carcinogenicity (Chu et al., 1981a; Larson et al., 1979a).

Choice of low-dose extrapolation approach

The MOA is a key consideration in clarifying how risks should be estimated for low-dose exposure. A linear-low-dose extrapolation approach was used to estimate human carcinogenic risk associated with chlordecone exposure, in the absence of information to inform the dose-response at low doses. Due to the early mortality in some dose groups, methods which can reflect the influence of intercurrent mortality on tumor incidence rates are preferred. U.S. EPA has generally used the multistage-Weibull model in this type of situation because it incorporates the time at which death-with-tumor occurred; however, it is unknown how well this model or the linear low-dose extrapolation predicts low-dose risks for chlordecone. The selected model does not represent all possible models one might fit, and other models could conceivably be selected to yield different results consistent with the observed data, both higher and lower than those included in this assessment. The human equivalent oral slope factors estimated from the statistically significant increase in liver tumors ranged from 1 per mg/kg-day in female mice to 10 per mg/kg-day in male mice, a range of one order of magnitude.

Dose metric

Chlordecone is not metabolized in rats or mice; however, in humans the majority of chlordecone is converted into chlordecone alcohol. Both compounds are non-mutagenic in salmonella (see Section 4.5.1). No information exists to inform the relative liver carcinogenicity of chlordecone alcohol compared to chlordecone. Noncancer effects of chlordecone (including neurological and liver effects) do not appear to be dependent on which moiety is produced. Regardless, as chlordecone is not metabolized in rats and mice, the test species of the only cancer bioassay, the administered dose was used as the dose metric. It is unknown whether conversion to chlordecone alcohol would have any effect on chlordecone's carcinogenicity.

Cross-species scaling

An adjustment for cross-species scaling ($BW^{3/4}$) was applied to address toxicological equivalence of internal doses between each rodent species and humans, consistent with the 2005 *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a). Without evidence to the contrary, it is assumed that equal risks result from equivalent constant lifetime exposures.

Statistical uncertainty at the POD

Measures of statistical uncertainty require assuming that the underlying model and associated assumptions are valid for the data under consideration. For the multistage-Weibull model applied to the male mice data, there is a reasonably typical degree of uncertainty at the 10% extra incidence level (the POD for linear low-dose extrapolation). The lower bound on the BMD_{10} ($BMDL_{10}$) for hepatocellular carcinoma in male mice is approximately 1.5-fold lower than the BMD_{10} .

Choice of species/gender

The oral slope factor for chlordecone was quantified using the tumor incidence data for male mice, which were found to be more sensitive than female mice or female rats to the carcinogenicity of chlordecone. The oral slope factor calculated from male mice was 6–7 times higher than the slope factors calculated from female mice and female rats. Liver tumor incidence in the high-dose group of male rats was far less robust (7%) than the high-dose groups of female rats, female mice, or male mice which had liver tumor incidences of 22, 47, and 88%, respectively. As there is no information to inform which species or gender of animals would be most applicable to humans, the most sensitive group was selected for the basis of the oral slope factor. The human relevance of the observed liver tumors is unknown. However, data in occupationally exposed workers indicate chlordecone predominantly accumulates in the liver (Cohn et al., 1978). Additionally, similarities in liver effects have been shown in occupationally exposed workers and in experimental animals including hypertrophy, hepatomegaly, and proliferation of metabolic enzymes. Though the MOA for observed liver tumors in rodents is unknown, the evidence suggesting the liver as a target organ of chlordecone toxicity and the concordance of liver tumors across both sexes of rats and mice lends strength to the concern for human carcinogenic potential.

Human population variability

The extent of inter-individual variability or sensitivity to the potential carcinogenicity of chlordecone is unknown. There are no data exploring whether there is differential sensitivity to chlordecone carcinogenicity across life stages. This lack of understanding about potential susceptibility differences across exposed human populations thus represents a source of uncertainty. Humans are expected to be more heterogenous than laboratory animals, and this

variability is likely to be influenced by ongoing or background exposures, diseases, and biological processes.

6. MAJOR CONCLUSIONS IN THE CHARACTERIZATION OF HAZARD AND DOSE RESPONSE

6.1. HUMAN HAZARD POTENTIAL

Chlordecone was previously used as an insecticide to control agricultural pests, including slugs, snails, and fire ants. Chlordecone was first produced in the United States in the early 1950s; however, production in the United States ended in 1975 due to intoxication from industrial exposure in employees who worked at a chlordecone manufacturing plant. Its registration was cancelled in 1976. Chlordecone is very resistant to degradation in the environment. It is expected to adsorb to soil and to stick to suspended solids and sediments in water. Very small amounts of chlordecone will evaporate from soil or water surfaces, and chlordecone in the air is likely to be removed by deposition of particles. Chlordecone has a high potential for bioaccumulation in fish and other aquatic organisms.

Chlordecone is well absorbed following oral exposure. Once absorbed, it is widely distributed and eventually concentrates in the liver. It is metabolized by humans and some animal species to chlordecone alcohol. Glucuronide conjugates of chlordecone and chlordecone alcohol, as well as unconjugated chlordecone, are slowly excreted in the bile and eliminated in the feces. Fecal excretion is delayed by enterohepatic recirculation.

The primary noncancer health effects of oral exposure to chlordecone in humans and animals include liver effects, kidney lesions (only in animals), neurotoxicity, and male reproductive toxicity. Other reproductive effects (i.e., PVE, impaired reproductive success) and developmental effects have also been observed in laboratory animals; however, the doses required to elicit these effects were generally higher than those that resulted in liver and kidney effects, neurotoxicity, and/or male reproductive toxicity.

Liver enlargement developed in workers exposed to high levels of chlordecone for an intermediate exposure duration; however, evidence of significant liver toxicity was not found. Histological changes were observed in liver biopsy samples; however, these were characterized as nonadverse in nature. Similar changes in the liver were also demonstrated in laboratory animals, including increased liver size and weight, hepatocellular hypertrophy, proliferation of the SER, increased microsomal protein, CYP450 content, cytochrome c reductase activity, and microsomal enzyme activity. Chronic animal studies also demonstrated evidence of hepatotoxicity, including hepatocellular hypertrophy, hyperplasia, congestion, mild fatty change, focal necrosis, and occasional small nests of proliferated sinusoidal cells.

Neurological symptoms were also reported in workers exposed to high doses of chlordecone, including tremor, headache, irritability, poor recent memory, rapid random eye movements, muscle weakness, gait ataxia, incoordination, and slurred speech. The effects persisted for as long as 9–10 months after cessation of exposure and the start of treatment.

Chlordecone also causes tremors, decreased motor coordination, hyperexcitability, and an exaggerated startle response in laboratory animals.

Chlordecone exposure in humans caused oligospermia, reduced sperm motility, and decreased libido in a group of men who were occupationally exposed to chlordecone for periods of up to 1.5 years. Upon re-examination of workers 5–7 years following the cessation of chlordecone exposure and treatment with cholestyramine, male reproductive parameters had returned to normal. Chlordecone also induces reproductive toxicity in male and female laboratory animals, as demonstrated by altered sperm parameters, testicular atrophy, altered estrous cyclicity, and impaired reproductive success.

Kidney toxicity was reported in laboratory animals but was not observed in occupationally exposed pesticide workers. However, it is unclear if clinical indicators of renal damage were specifically examined in occupationally exposed workers or whether signs of kidney impairment would be expected following the relatively short (5–6 month) average exposure durations. Several animal studies reported kidney effects from chlordecone exposure. Proteinuria and increased incidence of kidney lesions were observed in female Wistar rats and in (NZB × NZW)_{F1} mice. Chronic kidney inflammation was observed in male and female Osborne-Mendel rats. Twenty-eight days of dietary exposure to chlordecone produced eosinophilic inclusions in proximal tubules in male Sprague-Dawley rats. Most of the effects of chlordecone are thought to be produced by the parent compound, primarily by interfering with the function of mitochondrial and cellular membranes. Disruption of cellular homeostasis and energy production within the cell eventually leads to impaired cellular function. In the central nervous system, altered calcium homeostasis leads to changes in neurotransmitter activity. In the liver, membrane perturbation and inhibition of transport proteins at the bile canalicular membrane are thought to be related to chlordecone-induced hepatobiliary dysfunction. The reproductive and developmental effects of chlordecone are most likely related to endocrine disruption. Chlordecone exhibits estrogenic properties that may be related to impaired reproductive success and adverse effects on sperm.

There are no reports of cancer in humans associated with exposure to chlordecone. Increased incidence of hepatocellular carcinoma was observed in rats and mice following oral exposure to chlordecone (NCI, 1976a, b). Significantly increased incidence of hepatocellular carcinoma was observed in both sexes of mice compared to matched controls. The incidence of liver tumors in male and female rats was comparatively less robust but did reach statistical significance when compared to pooled laboratory controls.

6.2. DOSE RESPONSE

6.2.1. Noncancer

No studies on the toxicity of chlordecone following inhalation exposure in humans or laboratory animals were located. This lack of data precludes the derivation of the RfC.

The database for chlordecone includes limited human data from observational studies of occupationally exposed workers. The database also includes several studies in laboratory animals, including chronic and subchronic dietary exposure studies, and several subchronic studies with a wide variety of tissues and endpoints assessed. The database also includes several reproductive and developmental studies, including one study specifically assessing developmental neurotoxicity. Endpoints associated with oral exposure to chlordecone include lesions in the liver, kidney, and testis; neurological effects (i.e., tremors); and reduced fertility. Support for these endpoints exists across a range of diverse studies.

The observation of kidney, liver, and testicular effects in the principal study at similar dose levels creates some uncertainty in the selection of a critical effect that would be most appropriate in a chronic low-dose human exposure paradigm. Additionally, LOAELs but no NOAELs exist for some effects, such as PVE observed in animals. This creates uncertainty as to where the threshold falls for this effect. The most sensitive effect observed from chronic dietary exposure to chlordecone is the increased incidence of kidney lesions in female Wistar rats (Larson et al., 1979a). Furthermore, several additional animal studies, in both rats and mice, support findings of kidney effects with chlordecone exposure (Sobel et al., 2006, 2005; Chetty et al., 1993c; Chu et al., 1981a; Chernoff and Rodgers, 1976; NCI, 1976b). In light of the weight of evidence for kidney, testicular, and liver lesions seen in the chlordecone animal literature (see Section 5.1.1), kidney lesions were deemed to be the most supported, biologically significant effect on which to base the RfD. Some uncertainty exists regarding the lack of observable effects on the kidney in humans. However, it is unknown whether the relatively short average exposure duration of workers (5–6 months) was sufficient for the development of detectable kidney impairment. Additionally, it is unclear from the literature whether clinical tests sensitive to early kidney impairment were administered to exposed workers.

A high background percentage of kidney lesions (55%) was noted in the untreated males in the principal study (Larson et al., 1979a), with a lower background percentage (12%) in female rats. It is possible that the high background occurrence of age-related kidney damage in the test species contributes to the observation of kidney lesions as the most sensitive dose-related effect following chlordecone exposure. It is uncertain whether kidney effects would be more sensitive than other effects of chlordecone in a species without this background disease process in the kidney. However, an age-related decline in the glomerular function in humans occurs as well regardless of underlying renal disease (e.g., hypertension) with the presence of androgens as a prominent risk factor in both rats and humans (Baylis, 1994). In the absence of data to the

contrary, the MOA of increased kidney lesions observed in rats following chlordecone exposure are considered relevant to humans.

After consideration of all potential PODs, the RfD of 3×10^{-4} mg/kg-day was based on the increased incidence of kidney lesions in female Wistar rats, following chronic dietary administration of chlordecone (Larson et al., 1979a). To derive the RfD, the uncertainty factor approach, following U.S. EPA practices (U.S. EPA, 2002), was applied to the POD determined through BMD modeling of the critical effect of kidney lesions in female rats. Factors to account for uncertainties associated with the extrapolation from the POD derived from an animal study to a diverse human population of varying susceptibilities were applied. This extrapolation was accomplished through the application of default UFs due to limitations in the chlordecone database that precluded the derivation of chemical-specific adjustment factors.

The choice of BMD model is not expected to introduce a considerable amount of uncertainty in the risk assessment since the chosen response rate of 10% additional risk is within the observable range of the data. Furthermore, the ratio of the BMD to the BMDL for the model that best describes the incidence data for the critical effect is less than a factor of two, indicating a typical level of experimental variability.

The default UF of 10 for the extrapolation from animals to humans is a composite of uncertainty to account for toxicokinetic differences and TD differences between the animal species in which the POD was derived and humans. PBTK models can be useful for the evaluation of interspecies toxicokinetics; however, the chlordecone database lacks an adequate model that would inform potential differences. Data from workers occupationally exposed to chlordecone provide some information on the absorption, distribution, metabolism, and elimination of chlordecone in humans and indicate qualitatively that the toxicokinetics of chlordecone are similar between humans and animals. Additionally, biological effects, including neurological, hepatic, and reproductive effects, observed in animals and humans are similar in nature, indicating qualitatively similar TDs (though quantitative differences are less known). However, the magnitude of the similarities or differences in toxicokinetic and TD parameters cannot be calculated due to uncertainties regarding routes of exposure and doses for the occupationally exposed workers. Therefore, an UF of 10 to account for interspecies differences was used.

Limited data exist on effects of chlordecone in a small population of occupationally exposed workers. Some information from occupational exposure studies indicate a wide range of chlordecone blood levels (0.009–11.8 ppm; median of 1.8 ppm) in workers categorized as affected (Cannon et al., 1978). This large range of blood levels in symptomatic workers may reflect a high level of variability in response to chlordecone. Alternatively, it may also be partially explained by the authors' inclusive case definition as any worker reporting nervousness with or without objective neurological abnormalities (tremulousness, gait or motor abnormalities) upon examination. Workers with subjective symptoms alone represented 36% of

identified cases. Since potential variability in responses to chlordecone in the greater human population is unknown, the default uncertainty factor of 10 for intrahuman variability was not reduced. Human variation may be larger or smaller; however, chlordecone-specific data to examine the potential magnitude of human variability of response are unknown.

Uncertainties associated with data gaps in the chlordecone database have been identified. Data more fully characterizing potential multigenerational reproductive effects are lacking. Several one-generational reproductive studies have indicated decreased reproductive success in chlordecone-treated animals (Cannon and Kimbrough, 1979; Good et al., 1965; Huber et al., 1965). In addition, two nonstandard multigenerational studies exist that evaluate reproductive success of chlordecone-treated animals (Gellert and Wilson, 1979; Good et al., 1965). However, due to limited scope and design, these studies are not considered adequate for the assessment of multigenerational reproductive toxicity. Therefore, for the above data gaps in the chlordecone database, an UF of 3 was applied to the POD in the derivation of the RfD.

The overall confidence in the RfD and the principal study (Larson et al., 1979a) is medium. The principal study involves a sufficient number of animals per group, several dose levels, and a wide range of tissues and endpoints assessed. Confidence in the database is medium. The chlordecone database includes case studies of occupationally exposed workers, chronic and subchronic dietary exposure studies in laboratory animals, and several subchronic reproductive and developmental studies, including one developmental neurotoxicity study. However, the database is lacking a multigenerational reproductive toxicity study. Therefore, reflecting medium confidence in both the database and the principal study, confidence in the RfD is medium.

6.2.2. Cancer

Under the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), the database for chlordecone indicates that it is “likely to be carcinogenic to humans”. This determination is primarily based on the NCI (1976a, b) study, which found positive evidence of liver tumors in both sexes of rats and mice after chronic chlordecone dietary exposure. Additionally, data on mirex, a structurally similar chemical, also demonstrates an increase in hepatocellular adenomas or carcinomas in both sexes of rats and mice. However, unlike the observed cancer effects, some but not all of the noncancer effects noted for these two chemicals are similar as described in Section 4.5.3. This weight of evidence conclusion collectively takes into consideration the NCI (1976a, b) cancer bioassay, the available human studies, and other chronic animal bioassays.

The increased incidence of hepatocellular carcinoma in both sexes of rats and mice observed in the NCI (1976a, b) 80 week dietary study was used to calculate the oral slope factor for chlordecone. Due to high toxicity and resulting early deaths in some dose groups, methods which account for the influence of early mortality on tumor incidence rates were utilized. In this case, the multistage-Weibull model was used because it incorporates the time at which death-

with-tumor occurred. An oral slope factor based on the incidence of hepatocellular carcinomas for each species/sex combination was calculated. Based on the modeling results (see Table 5-5), male mice are the most sensitive to tumor induction following exposure to chlordecone, and were thus used as the basis of the oral slope factor. The oral slope factor was derived from the BMDL₁₀, the 95% lower bound on the dose associated with a 10% extra cancer risk of hepatocellular carcinoma in male B6C3F₁ mice, by dividing the BMR (0.10) by the BMDL₁₀, and represents an upper bound, continuous lifetime exposure estimate of cancer potency. The BMDL₁₀, lower 95% bound on exposure at 10% extra risk, is 1.01×10^{-2} mg/kg-day and the slope of the linear extrapolation from the BMDL₁₀ to 0 = $0.10/1.01 \times 10^{-2} = 10$ (mg/kg-day)⁻¹. Therefore, the recommended oral cancer slope factor for use in estimating human cancer risk from continuous lifetime oral exposure to chlordecone is 10 per mg/kg-day.

Areas of uncertainty exist for this cancer assessment. The multistage-Weibull model was selected to model liver tumor incidence in rats and mice; however, it is unknown how well this model or the linear low-dose extrapolation predicts low-dose risks for chlordecone. The selected model does not represent all possible models one might fit, and other models could conceivably be selected to yield different results consistent with the observed data, both higher and lower than those included in this assessment. The human equivalent oral slope factors estimated from the statistically significant increase in liver tumors ranged from 1.4 per mg/kg-day in female mice to 9.9 per mg/kg-day in male mice, a range of one order of magnitude. The oral slope factor for chlordecone was quantified using the tumor incidence data for male mice, which were found to be more sensitive than female mice or female rats. The oral slope factor calculated from male mice was 6-7 times higher than the slope factors calculated from female mice and female rats. As there is no information to inform which species or gender of animals would be most applicable to humans, the most sensitive group was selected for the basis of the oral slope factor. The human relevance of the observed tumors is unknown; however, in the absence of MOA data indicating liver tumors observed in rats and mice with chlordecone exposure would not be expected to occur in humans, these tumors are considered to be relevant to humans.

7. REFERENCES

- Adir, J; Caplan, YH; Thompson, BC. (1978) Kepone serum half-life in humans. *Life Sci* 22(8):699–702.
- Ahlborg, UG; Lipworth, L; Titus-Ernstoff, L; et al. (1995) Organochlorine compounds in relation to breast cancer, endometrial cancer, and endometriosis: an assessment of the biological and epidemiological evidence. *CRC Crit Rev Toxicol* 25(6):463–531.
- Ahmed, SA; Hissong, BD; Verthelyi, D; et al. (1999) Gender and risk of autoimmune diseases: possible role of estrogenic compounds. *Environ Health Perspect* 107(Suppl. 5):681–686.
- Aldous, CN; Chetty, CS; Desai, D. (1983) Alterations in tissue distribution of chlordecone (Kepone) in the rat following phenobarbital or SKF-525A administration. *J Toxicol Environ Health* 11(3):365–372.
- ATSDR (Agency for Toxic Substances and Disease Registry). (1995) Toxicological profile for mirex and chlordecone. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA. Available online at <http://www.atsdr.cdc.gov/toxpro2.html> (accessed June 25, 2009).
- Bale, SS. (1983) Cytological effects of Kepone on Chinese hamster cells. *J Hered* 74(2):123–124.
- Baylis, C. (1994) Age-dependant glomerular damage in the rat. *J Clin Invest* 94:1823–1829.
- Belfiore, CJ; Yang, RS; Chubb, LS; et al. (2007) Hepatic sequestration of chlordecone and hexafluoroacetone evaluated by pharmacokinetic modeling. *Toxicology* 234(1–2):59–72.
- Blain, RB; Reeves, R; Ewald, KA; et al. (1999) Susceptibility to chlordecone-carbon tetrachloride induced hepatotoxicity and lethality is both age and sex dependent. *Toxicol Sci* 50(2):280–286.
- Blanke, RV; Fariss, MW; Guzelian, PS; et al. (1978) Identification of a reduced form of chlordecone (Kepone) in human stool. *Bull Environ Contam Toxicol* 20:782–785.
- Bolger, R; Wiese, TE; Ervin, K; et al. (1998) Rapid screening of environmental chemicals for estrogen receptor binding capacity. *Environ Health Perspect* 106(9):551–557.
- Bondy, SC; McKee, M. (1990) Prevention of chemically induced synaptosomal changes. *J Neurosci Res* 25:229–235.
- Boylan, JJ; Cohn, WJ; Egle, JL, Jr; et al. (1979) Excretion of chlordecone by the gastrointestinal tract: evidence for a nonbiliary mechanism. *Clin Pharmacol Ther* 25:579–585.
- Brown, HE; Salamance, S; Stewart, G; et al. (1991) Chlordecone (Kepone) on the night of proestrus inhibits female sexual behavior in CDF-344 rats. *Toxicol Appl Pharmacol* 110(1):97–106.
- Bulger, WH; Muccitelli, RM; Kupfer, D. (1979) Studies on the estrogenic activity of chlordecone (Kepone) in the rat: effects on uterine estrogen receptor. *Mol Pharmacol* 15:515–524.
- Bungay, PM; Dedrick, RL; Matthews, HB. (1979) Pharmacokinetics of halogenated hydrocarbons. *Ann NY Acad Sci* 320:257–270.
- Bus, JS; Leber, AP. (2001) Miscellaneous chlorinated hydrocarbon pesticides. In: Bingham, E; Cohn, B; Powell, CH; eds. *Patty's toxicology*. Electronic version available through subscription to Wiley Interscience. Available online at http://www.mrw.interscience.wiley.com/pattys/pattys_search_fs.html (accessed March 24, 2004).
- Cai, Z; Mehendale, HM. (1993) Resiliency to amplification of carbon tetrachloride hepatotoxicity by chlordecone during postnatal development in rats. *Pediatr Res* 33(3):225–232.

- Caldwell, V; Loch-Carus, R. (1992) Chlordecone rapidly and reversibly inhibits gap junctional communication in human embryonic palatal mesenchyme cells. *In Vitro Toxicol* 5(2):113–122.
- Cannon, SB; Kimbrough, RD. (1979) Short-term chlordecone toxicity in rats including effects on reproduction, pathological organ changes, and their reversibility. *Toxicol Pharmacol* 47:469–476.
- Cannon, SB; Veazey, JM; Jackson, RS; et al. (1978) Epidemic Kepone poisoning in chemical workers. *Am J Epidemiol* 107:529–537.
- Carpenter, HM; Curtis, LR. (1989) A characterization of chlordecone pretreatment-altered pharmacokinetics in mice. *Drug Metab Dispos* 17(2):131–138.
- Carpenter, HM; Curtis, LR. (1991) Low dose chlordecone pretreatment altered cholesterol disposition without induction of cytochrome P-450. *Drug Metab Dispos* 19(3):673–678.
- Carpenter, HM; Hedstrom, OR; Siddens, LK; et al. (1996) Ultrastructural, protein, and lipid changes in liver associated with chlordecone treatment of mice. *Fundam Appl Toxicol* 34(1):157–164.
- Chadwick, RW; Copeland, MF; Rosenstein, L. (1979) The effect of Kepone exposure during gestation and lactation on the metabolism of lindane by weanling rats. *Toxicol Lett* 4:247–252.
- Chernoff, N; Kavlock, RJ. (1982) An in vivo teratology screen utilizing pregnant mice. *J Toxicol Environ Health* 10:541–550.
- Chernoff, N; Rogers, EH. (1976) Fetal toxicity of Kepone in rats and mice. *Toxicol Appl Pharmacol* 38:189–194.
- Chetty, KN; Walker, J; Brown, K; et al. (1993a) The effects of dietary calcium and chlordecone on cholinesterase, triglycerides, low density lipoproteins, and cholesterol in serum of rat. *Arch Environ Contam Toxicol* 24(3):365–367.
- Chetty, KN; Brown, K; Walker, J; et al. (1993b) Effects of chlordecone and malnutrition on immune response in rats. *Life Sci* 52(18):PL175–180.
- Chetty, KN; Walker, J; Brown, K; et al. (1993c) Influence of dietary calcium on chlordecone-induced biochemical changes in serum of rat. *Ecotoxicol Environ Saf* 26(2):248–252.
- Chu, I; Villeneuve, DC; Becking, GC; et al. (1980) Short-term study of the combined effects of mirex, photomirex, and Kepone with halogenated biphenyls in rats. *J Toxicol Environ Health* 6:421–432.
- Chu, I; Villeneuve, DC; Valli, VE; et al. (1981a) Chronic toxicity of photomirex in the rat. *Toxicol Appl Pharmacol* 59:268–278.
- Chu, I; Villeneuve, DC; MacDonald, BL; et al. (1981b) Reversibility of the toxicological changes induced by photomirex and mirex. *Toxicology* 21(3):235–250.
- Chu, I; Villeneuve, DC; Secours, VE; et al. (1981c) Effects of photomirex and mirex on reproduction in the rat. *Toxicol Appl Pharmacol* 60(3):549–556.
- Cohn, WJ; Boylan, JJ; Blanke, RV; et al. (1978) Treatment of chlordecone (Kepone) toxicity with cholestyramine: results of a controlled trial. *N Engl J Med* 298:243–248.
- Cooper, JR; Vodcnik, MJ; Gordon JH. (1985) Effects of perinatal Kepone exposure on sexual differentiation of the rat brain. *Neurotoxicology* 6(1):183–190.
- Curtis LR, Mehendale HM. (1979) The effects of kepone pretreatment on biliary excretion of xenobiotics in the male rat. *Toxicol Appl Pharmacol* 47(2):295-303.

- Curtis, LR; Thureson-Klein, AK; Mehendale, HM. (1981) Ultrastructural and biochemical correlates of the specificity of chlordecone-potentiated carbon tetrachloride hepatotoxicity. *J Toxicol Environ Health* 7(3-4):499-517.
- Dalu, A; Warbritton, A; Bucci, TJ; et al. (1995) Age-related susceptibility to chlordecone-potentiated carbon tetrachloride hepatotoxicity and lethality is due to hepatic quiescence. *Pediatr Res* 38(2):140-148.
- Dalu, A; Rao, PS; Mehendale, HM. (1998) Colchicine antimetabolism abolishes resiliency of postnatally developing rats to chlordecone-amplified carbon tetrachloride hepatotoxicity and lethality. *Environ Health Perspect* 106(9):597-606.
- Das, SK; Taylor, JA; Korach, KS; et al. (1997) Estrogenic responses in estrogen receptor-alpha deficient mice reveal a distinct estrogen signaling pathway. *Proc Natl Acad Sci* 94:12786-12791.
- Deml E, Oesterle D. (1987) Dose-response of promotion by polychlorinated biphenyls and chloroform in rat liver foci bioassay. *Arch Toxicol* 60:209-211.
- Desaiah, D. (1982) Biochemical mechanisms of chlordecone neurotoxicity: a review. *Neurotoxicology* 3(2):103-110.
- Desaiah, D. (1985) Chlordecone interaction with catecholamine binding and uptake in rat brain synaptosomes. *Neurotoxicology* 6(1):159-165.
- Egle, JL, Jr.; Fernandez, JB; Guzelian, PS; et al. (1978) Distribution and excretion of chlordecone (Kepone) in the rat. *Drug Metab Dispos* 6(1):91-95.
- el-Masri, HA; Thomas, RS; Benjamin, SA; et al. (1995) Physiologically based pharmacokinetic/pharmacodynamic modeling of chemical mixtures and possible applications in risk assessment. *Toxicology* 105(2-3):275-282.
- End, DW; Carchman, RA; Dewey, WL. (1981) Neurochemical correlates of chlordecone neurotoxicity. *J Toxicol Environ Health* 8(5-6):707-718.
- Fariss, MW; Blanke, RV; Saady, V; et al. (1980) Demonstration of major metabolic pathways for chlordecone (Kepone) in humans. *Drug Metab Dispos* 8:434-438.
- Fujimori, K; Benet, H; Mehendale, HM; et al. (1982a) Comparison of brain discrete area distributions of chlordecone and mirex in the mouse. *Neurotoxicology* 3(2):125-129.
- Fujimori, K; Nabeshima, T; Ho, IK; et al. (1982b) Effects of oral administration of chlordecone and mirex on brain biogenic amines in mice. *Neurotoxicology* 3(2):143-148.
- Gaines TB. (1969). Acute toxicity of pesticides. *Toxicol Appl Pharmacol* 14:515-534.
- Gaines, TB; Kimbrough, RD. (1970) Oral toxicity of mirex in adult and suckling rats. With notes on the ultrastructure of liver changes. *Arch Environ Health* 21:7-14.
- Galloway, SM; Armstrong, MJ; Reuben, C; et al. (1987) Chromosome aberrations and sister chromatid exchanges in Chinese hamster ovary cells: evaluations of 108 chemicals. *Environ Mol Mutagen* 10(Suppl. 10):1-175.
- Gart, JJ; Krewski, D; Lee, PN; et al. (1986) Statistical methods in cancer research. Volume III: The design and analysis of long-term animal experiments. *IARC Sci Publ.* (79):1-219.
- Gellert, RJ; Wilson, C. (1979) Reproductive function in rats exposed prenatally to pesticides and polychlorinated biphenyls (PCB). *Environ Res* 18:437-443.
- Gibson, JR; Ivie, GW; Dorough, HW. (1972) Fate of mirex and its major photodecomposition product in rats. *J Agric Food Chem* 20(6):1246-1248.

- Gilroy, DJ; Carpenter, HM; Curtis, LR. (1994) Chlordecone pretreatment alters [14C]chlordecone and [14C]cholesterol transport kinetics in the perfused rat liver. *Fundam Appl Toxicol* 22(2):286–292.
- Good, EE; Ware, GW; Miller, DF. (1965) Effects of insecticides on reproduction in the laboratory mouse: I. Kepone. *J Econ Entomol* 58(4):754–757.
- Guzelian, PS. (1982a) Chlordecone poisoning: a case study in approaches for the detoxification of humans exposed to environmental chemicals. *Drug Metab Rev* 13:663–679.
- Guzelian, PS. (1982b) Comparative toxicology of chlordecone (Kepone) in humans and experimental animals. *Ann Rev Pharmacol Toxicol* 22:89–113.
- Guzelian, PS; Vranian, G; Boylan, JJ; et al. (1980) Liver structure and function in patients poisoned with chlordecone (Kepone). *Gastroenterology* 78(2):206–213.
- Hammond B; Katzyellenbogen BS; Krauthammer N; et al. (1979). Estrogenic activity of the insecticide chlordecone (Kepone) and interaction with uterine estrogen receptors. *Proc Natl Acad Sci* 76:6641–6645.
- Hansch, C; Leo, A; Hoekman, D. (1995) Exploring QSAR: hydrophobic, electronic, and steric constants. ACS Professional Reference Book. Washington, DC: American Chemical Society; p. 65.
- Hart, SE; Kinter, LB. (2005). Assessing renal effects of toxicants in vivo. In: Tarloff, JB; Lawrence, LH; eds. *Toxicology of the kidney*. New York, NY: CRC Press; pp. 81–147.
- Heatherington, AC; Fisher, HL; Sumler, MR; et al. (1998) Percutaneous absorption and disposition of [14C]chlordecone in young and adult female rats. *Environ Res* 79(2):138–155.
- Herr DW, Gallus JA, Tilson HA. (1987) Pharmacological modification of tremor and enhanced acoustic startle by chlordecone and p,p'-DDT. *Psychopharmacology*. 91(3):320-5.
- Hewitt, LA; Caille, G; Plaa, GL; et al. (1985) Temporal relationships between biotransformation, detoxication, and chlordecone potentiation of chloroform-induced hepatotoxicity. *Can J Physiol Pharmacol* 64:477–482.
- Hodges LC, Bergerson JS, Hunter DS, et al. (2000). Estrogenic effects of organochlorine pesticides on uterine leiomyoma cells in vitro. *Toxicol Sci*. 54(2):355–364.
- Hoskins B, Ho IK. (1982) Chlordecone-induced alterations in content and subcellular distribution of calcium in mouse brain. *J Toxicol Environ Health*. 9(4):535–44.
- Houston, TE; Mutter, LC; Blanke, RV; et al. (1981) Chlordecone alcohol formation in the Mongolian gerbil (*Meriones unguiculatus*): A model for human metabolism of chlordecone (Kepone). *Fundam Appl Toxicol* 1(3):293–298.
- Huang, ESR; Nelson, FR. (1986) Anti-estrogenic action of chlordecone in rat pituitary gonadotrophs in vitro. *Toxicol Appl Pharmacol* 82:62–69.
- Huang, TP; Ho, IK; Mehendale, HM. (1980) Assessment of neurotoxicity induced by oral administration of chlordecone (Kepone) in the mouse. *Neurotoxicology* 2:113–124.
- Huber, JJ. (1965) Some physiological effects of the insecticide Kepone in the laboratory mouse. *Toxicol Appl Pharmacol* 7:516–524.
- IARC (International Agency for Research on Cancer). (1979) Chlordecone. IARC monographs on the evaluation of carcinogenic risk of chemicals to humans. Vol. 20. Some halogenated hydrocarbons. Lyon, France: International Agency for Research on Cancer; p. 67.
- Innes, JRM; Ulland, BM; Valerio, MG; et al. (1969) Bioassay of pesticides and industrial chemicals for tumorigenicity in mice: a preliminary note. *J Natl Cancer Inst* 42(6):1101–1114.

- Ivie, GW; Gibson, JR; Bryant, HE; et al. (1974) Accumulation, distribution, and excretion of mirex-¹⁴C in animals exposed for long periods to the insecticide in the diet. *J Agric Food Chem* 22(4):646–653.
- Jinna, RR; Uzodinma, JE; Desai, D. (1989) Age-related changes in rat brain ATPases during treatment with chlordecone. *J Toxicol Environ Health* 27(2):199–208.
- Johnson, DC. (1996) Estradiol-chlordecone (Kepone) interactions: additive effect of combinations for uterotrophic and embryo implantation functions. *Toxicol Lett* 89:57–64.
- Johnson, DC; Sen, M; Kogo, H; et al. (1990) Initiation of embryo implantation and maintenance of early pregnancy in the rat by chlordecone (Kepone). *Proc Soc Exp Biol Med* 195(1):44–50.
- Johnson, DC; Banerjee, S; Chatterjee, S. (1995) Estradiol and chlordecone (Kepone) decrease adenosine 3'5'-cyclic monophosphate concentrations in the ovariectomized immature rat uterus. *Proc Soc Exp Biol Med* 210(1):33–38.
- Kavlock, RJ; Chemoff, N; Rogers, E; et al. (1980) Comparative tissue distribution of mirex and chlordecone in fetal and neonatal rats. *Pestic Biochem Physiol* 14(3):227–235.
- Kennedy, MW; Pittman, KA; Stein, VM. (1975) Fate of ¹⁴C mirex in the female rhesus monkey. *Toxicol Appl Pharmacol* 33:161–162.
- Kilzer, L; Scheunert, I; Geyer, H; et al. (1979) Laboratory screening of the volatilization rates of organic chemicals from water and soil. *Chemosphere* 10:751–761.
- Kitchin, KT; Brown, JL. (1989) Biochemical studies of promoters of carcinogenesis in rat liver. *Teratogen Carcinogen Mutagen* 9:273–285.
- Kocarek, TA; Schuetz, EG; Guzelian, PS. (1991) Selective induction of cytochrome P450e by Kepone (chlordecone) in primary cultures of adult rat hepatocytes. *Mol Pharmacol* 40:203–210.
- Kocarek, TA; Schuetz, EG; Guzelian, PS. (1994) Regulation of cytochrome P450 2B1/2 mRNAs by Kepone (chlordecone) and potent estrogens in primary cultures of adult rat hepatocytes on Matrigel. *Toxicol Lett* 71(2):183–196.
- Kodavanti, PR; Joshi, UM; Mehendale, HM; et al. (1989) Chlordecone (Kepone)-potentiated carbon tetrachloride hepatotoxicity in partially hepatectomized rats: a histomorphometric study. *J Appl Toxicol* 9(6):367–375.
- Kodavanti, PR; Kodavanti, UP; Mehendale, HM. (1990) Altered hepatic energy status in chlordecone (Kepone)-potentiated CCl₄ hepatotoxicity. *Biochem Pharmacol* 40(4):859–866.
- Kodavanti, PRA; Kodavanti, UP; Faroon, OM; et al. (1992) Pivotal role of hepatocellular regeneration in the ultimate hepatotoxicity of CCl₄ in chlordecone-, mirex-, or phenobarbital-pretreated rats. *Toxicol Pathol* 20(4):556–569.
- Kodavanti, PRA; Rao, VC; Mehendale, HM. (1993) Loss of calcium homeostasis leads to progressive phase of chlordecone-potentiated carbon tetrachloride hepatotoxicity. *Toxicol Appl Pharmacol* 122:77–87.
- Kuiper, GGJM; Lemmen, JG; Carlsson, B; et al. (1998) Interaction of estrogenic chemicals and phytoestrogens with estrogen receptor beta. *Endocrinology* 139(10):4252–4263.
- Krewski, D; Crump, KS; Farmer, J; et al. (1983) A comparison of statistical methods for low dose extrapolation utilizing time-to-tumor data. *Fundam Appl Toxicol* 3:140–160.
- Laessig, SA; Auger, AP; McCarthy, MM; et al. (2007) Effects of prenatal chlordecone on sexually differentiated behavior in adult rats. *Neurotoxicol Teratol* 29(2):255–263.
- Lahita, RG. (1997) Predisposing factors to autoimmune disease. *Int J Fertil Womens Med* 42(2):115–119.

- Larson, PS; Egle, JL, Jr; Hennigar, GR. (1979a) Acute, subchronic, and chronic toxicity of chlordecone. *Toxicol Appl Pharmacol* 48:29–41.
- Larson, PS; Egle, JL, Jr; Hennigar, GR; et al. (1979b) Acute and subchronic toxicity of mirex in the rat, dog and rabbit. *Toxicol Appl Pharmacol* 49(2):271–277.
- Lide, DR; ed. (2000) *CRC handbook of chemistry and physics*. 81st edition. Boca Raton, FL: CRC Press; pp. 3–204.
- Linder, RE; Scotti, TM; McElroy, WK; et al. (1983) Spermotoxicity and tissue accumulation of chlordecone (Kepone) in male rats. *J Toxicol Environ Health* 12:183–192.
- Mactutus, CF; Tilson, HA. (1985) Evaluation of long-term consequences in behavioral and/or neural function following neonatal chlordecone exposure. *Teratology* 31(2):177–186.
- Martinez, AJ; Taylor, JR; Dyck, PJ; et al. (1978) Chlordecone intoxication in man: II. Ultrastructure of peripheral nerves and skeletal muscle. *Neurology* 28:631–635.
- Medical College of Wisconsin. (1999) Glomerulosclerosis. Healthlink. Medical College of Wisconsin. Available online at: <http://healthlink.mcw.edu/article/943054092.html> (accessed June 25, 2009).
- Mehendale, HM. (1990) Potentiation of halomethane hepatotoxicity by chlordecone: a hypothesis for the mechanism. *Med Hypotheses* 33(4):289–299.
- Mehendale, HM. (1994) Amplified interactive toxicity of chemicals at nontoxic levels: mechanistic considerations and implications to public health. *Environ Health Perspect* 102(Suppl. 9):139–149.
- Mehendale, HM; Klingensmith, JS. (1988) In vivo metabolism of CCl₄ by rats pretreated with chlordecone, mirex, or phenobarbital. *Toxicol Appl Pharmacol* 93(2):247–256.
- Mehendale, HM; Takanaka, A; Desai, D; et al. (1977) Kepone induction of hepatic mixed function oxidases. *Life Sci* 20(6):991–997.
- Mehendale, HM; Takanaka, A; Desai, D; et al. (1978) Effect of preexposure to Kepone on hepatic mixed function oxidases in the female rat. *Toxicol Appl Pharmacol* 44:171–180.
- Mehendale, HM; Purushotham, KR; Lockard, VG. (1989) The time course of liver injury and [3H]thymidine incorporation in chlordecone-potentiated CHCl₃ hepatotoxicity. *Exp Mol Pathol* 51:31–47.
- Metcalf, RL. (2002) Insect control. In: *Ullmann's encyclopedia of industrial chemistry*. Wiley Interscience. Available online at http://www.mrw.interscience.wiley.com/ueic/ueic_search_fs.html (accessed March 22, 2004).
- Mitra, A; Richards, I; Kitchin, K; et al. (1990) Mirex induces ornithine decarboxylase in female rat liver. *J Biochem Toxicol* 5(2):119–124.
- Molowa, DT; Wrighton, SA; Blanke, RV; et al. (1986) Characterization of a unique aldo-keto reductase responsible for the reduction of chlordecone in the liver of the gerbil and man. *J Toxicol Environ Health* 17:375–384.
- Morgan DP, Sandifier SH, Hetzler HL et al; (1979) Test for in vivo conversion of mirex to kepone. *Bull Environ Contam Toxicol* 22(1-2):238-44.
- Mortelmans, K; Haworth, S; Lawlor, T; et al. (1986) Salmonella mutagenicity tests: II. Results from the testing of 270 chemicals. *Environ Mutagen* 8(Suppl. 7):1–119.
- Murali, B; Korrapati, MC; Anand, SS; et al. (2002) Age-dependent susceptibility of F344 rats to chlordecone potentiated CCl₄ hepatotoxicity and lethality. *Int J Toxicol* 21(6):524.

- NCI (National Cancer Institute). (1976a) Report on carcinogenesis bioassay of technical grade chlordecone (Kepone) (CAS No. 143-50-0). Carcinogenesis Program, Division of Cancer Cause and Prevention, National Cancer Institute, Public Health Service, U.S. Department of Health and Human Services; NTP TR-00. Available online at <http://ntp.niehs.nih.gov/> (accessed June 25, 2009).
- NCI (National Cancer Institute). (1976b) Carcinogenesis bioassay of technical grade chlordecone (Kepone) (CAS No. 143-50-0). Project # 455-5224. (unpublished raw data).
- NIOSH (National Institute for Occupational Safety and Health). (2004) Pocket guide to chemical hazards. Centers for Disease Control and Prevention, Public Health Service, U.S. Department of Health and Human Services, Cincinnati, Ohio. Available online at <http://www.cdc.gov/niosh/npg/> (accessed March 23, 2004).
- NLM (National Library of Medicine). (2004a) Chlordecone. HSDB (Hazardous Substances Data Bank). National Institutes of Health, U.S. Department of Health and Human Services, Bethesda, Maryland. Last review dated August 25, 1989. Available online at <http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB> (accessed June 25, 2009).
- NLM (National Library of Medicine). (2004b) Chlordecone. ChemIDplus. National Institutes of Health, U.S. Department of Health and Human Services, Bethesda, Maryland. Available online at <http://chem.sis.nlm.nih.gov/chemidplus/jsp/common/ChemFull.jsp?MW=490.639> (accessed February 6, 2007).
- NLM. (National Library of Medicine). (2004c) Mirex. ChemIDplus. National Institutes of Health, U.S. Department of Health and Human Services, Bethesda, Maryland. Available online at <http://chem.sis.nlm.nih.gov/chemidplus/jsp/common/ChemFull.jsp?MW=545.546> (accessed February 6, 2007).
- NRC (National Research Council). (1983) Risk assessment in the federal government: managing the process. Washington, DC: National Academy Press.
- NTP (National Toxicology Program). (1990) Toxicology and carcinogenesis studies of mirex (CAS No. 2385-85-5) in F344/N rats (feed studies). Public Health Service, U.S. Department of Health and Human Services; NTP TR-313; NIH Publ. No. 90-2569. Available from the National Institute of Environmental Health Sciences, Research Triangle Park, NC, and online at <http://ntp.niehs.nih.gov/ntpweb/index.cfm?objectid=08480D3B-9D26-6528-FDD1F546F54EE341> (accessed June 25, 2009).
- O'Neil, MJ; ed. (2001) The Merck index: an encyclopedia of chemicals, drugs, and biologicals. 13th edition. Whitehouse Station, NJ: Merck and Co., Inc.; p. 359.
- Pinkston, G; Uphouse, L. (1987–1988) Postovulatory reduction of fertility in chlordecone treated female rats. *Reprod Toxicol* 1(2):105–109.
- Pittman, KA; Wiener, W; Treble, DH. (1976) Mirex kinetics in the rhesus monkey. II. Pharmacokinetic model. *Drug Metab Dispos* 4(3):288–295.
- Plaa, GL; Caille, G; Vezina, M; et al. (1987) Chloroform interaction with chlordecone and mirex: correlation between biochemical and histological indexes of toxicity and quantitative tissue levels. *Fundam Appl Toxicol* 9:198–207.
- Probst, GS; McMahon, RE; Hill, LE; et al. (1981) Chemically-induced unscheduled DNA synthesis in primary rat hepatocyte cultures: a comparison with bacterial mutagenicity using 218 compounds. *Environ Mutagen* 3:11–32.
- Pryor GT, Uyeno ET, Tilson HA, et al.; (1983) Assessment of chemicals using a battery of neurobehavioral tests: a comparative study. *Neurobehav Toxicol Teratol*. 5(1):91-117.
- PWG (Pathology Working Group). (1992) Pathology Working Group report on mirex chronic toxicity/carcinogenicity study in F344 rats. Prepared by R.M. Sauer, PATHCO, Inc. Ijamsville, MD. (unpublished report).
- Rao, SB; Mehendale, HM. (1989) Protection from chlordecone (Kepone)-potentiated CCl₄ hepatotoxicity in rats by fructose 1,6-diphosphate. *Int J Biochem* 21(9):949–954.

- Rao, SB; Young, RA; Mehendale, HM. (1990) Perturbations in polyamines and related enzymes following chlordecone-potentiated bromotrichloromethane hepatotoxicity. *J Biochem Toxicol* 5(1):23–32.
- Rochelle, LG; Curtis LR. (1994) Distribution of chlordecone to liver plasma membranes and recovery from hepatobiliary dysfunction in rats. *Toxicology* 86(1–2):123–134.
- Rochelle, LG; Miller, TL; Curtis, LR. (1990) Chlordecone impairs Na(+)-stimulated L-[3H]glutamate transport and mobility of 16-doxyl stearate in rat liver plasma membrane vesicles. *Toxicol Appl Pharmacol* 105(2):234–242.
- Rosecrans, JA; Squibb, RE, Jr; Johnson, JH; et al. (1985) Effects of neonatal chlordecone exposure on pituitary-adrenal function in adult Fischer-344 rats. *Neurobehav Toxicol Teratol* 7(1):33–37.
- Sanborn, GE; Selhorst, JB; Calabrese, VP; et al. (1979) *Pseudotumor cerebri* and insecticide intoxication. *Neurology* 29(9 Pt 1):1222–1227.
- Schoeny, RS; Smith, CS; Loper, JC. (1979) Non-mutagenicity for salmonella of the chlorinated hydrocarbons aroclor 1254, 1,2,4-trichlorobenzene, mirex and Kepone. *Mutat Res* 68:125–132.
- Schrader, TJ; Cooke, GM. (2000) Examination of selected food additives and organochlorine food contaminants for androgenic activity in vitro. *Toxicol Sci* 53:278–288.
- Scippo, ML; Argiris, C; Van De Weerd, C; et al. (2004) Recombinant human estrogen, androgen and progesterone receptors for detection of potential endocrine disruptors. *Anal Bioanal Chem* 378:664–669.
- Seidenberg, JM; Anderson, DG; Becker, RA. (1986) Validation of an in vivo developmental toxicity screen in the mouse. *Teratogen Carcinogen Mutagen* 6:361–374.
- Shah, PV; Fisher, HL; Sumler, MR; et al. (1987) Comparison of the penetration of 14 pesticides through the skin of young and adult rats. *J Toxicol Environ Health* 21(3):353–366.
- Sierra, V; Uphouse, L. (1986) Long-term consequences of neonatal exposure to chlordecone. *Neurotoxicology* 7(2):609–621.
- Simon, GS; Kipps, BR; Tardiff, RG; et al. (1978) Failure of Kepone and hexachlorobenzene to induce dominant lethal mutations in the rat. *Toxicol Appl Pharmacol* 45(1):330–331.
- Simon, GS; Egle, JL, Jr; Dougherty, RW; et al. (1986) Dominant lethal assay of chlordecone and its distribution in the male reproductive tissues of the rat. *Toxicol Lett* 30:237–245.
- Sirica, AE; Wilkerson, CS; Wu, LL; et al. (1989) Evaluation of chlordecone in a two-stage model of hepatocarcinogenesis: a significant sex difference in the hepatocellular carcinoma incidence. *Carcinogenesis* 10(6):1047–1054.
- Skalsky, HL; Farris, MW; Blanke, RV; et al. (1979) The role of plasma proteins in the transport and distribution of chlordecone (Kepone) and other polyhalogenated hydrocarbons. *Ann NY Acad Sci* 320:231–237.
- Smialowicz, RJ; Luebke, RW; Riddle, MM; et al. (1985) Evaluation of the immunotoxic potential of chlordecone with comparison to cyclophosphamide. *J Toxicol Environ Health* 15(5):561–574.
- Sobel, ES; Gianini, J; Butfiloski, EJ; et al. (2005) Acceleration of autoimmunity by organochlorine pesticides in (NZB × NZW)F₁ mice. *Environ Health Perspect* 113(3):323–328.
- Sobel, ES; Wang, F; Butfiloski, EJ; et al. (2006) Comparison of chlordecone effects on autoimmunity in (NZB × NZW) F(1) and BALB/c mice. *Toxicology* 218(2–3):81–89.
- Soine, PJ; Blanke, RV; Guzelian, PS; et al. (1982) Preferential binding of chlordecone to the protein and high density lipoprotein fractions of plasma from humans and other species. *J Toxicol Environ Health* 9:107–118.
- Soine, PJ; Blanke, RV; Schwartz, CC. (1983) Chlordecone metabolism in the pig. *Toxicol Lett* 17(1–2):35–41.

- Soine, PJ; Blanke, RV; Chinchilli, VM; et al. (1984) High-density lipoproteins decrease the biliary concentration of chlordecone in isolated perfused pig liver. *J Toxicol Environ Health* 14(2-3):319-335.
- Soni, MG; Mehendale, HM. (1991a) ATP protection of chlordecone-amplified carbon tetrachloride hepatotoxicity and lethality. *Fed Am Soc Exp Biol J* 5(6):A1604.
- Soni, MG; Mehendale, HM. (1991b) Protection from chlordecone-amplified carbon tetrachloride toxicity by cyanidanol: regeneration studies. *Toxicol Appl Pharmacol* 108(1):58-66.
- Soni, MG; Mehendale, HM. (1991c) Protection from chlordecone-amplified carbon tetrachloride toxicity by cyanidanol: biochemical and histological studies. *Toxicol Appl Pharmacol* 108(1):46-57.
- Soni, MG; Mehendale, HM. (1993) Hepatic failure leads to lethality of chlordecone-amplified hepatotoxicity of carbon tetrachloride. *Fundam Appl Toxicol* 21(4):442-450.
- Squibb, RE; Tilson, HA. (1982) Effects of gestational and perinatal exposure to chlordecone (Kepone) on the neurobehavioral development of Fischer-344 rats. *Neurotoxicology* 3(2):17-26.
- Starcevic, SL; Bortolin, S; Woodcroft, KJ; et al. (2001) Kepone (chlordecone) disrupts adherens junctions in human breast epithelial cells cultured on Matrigel. *In Vivo* 15:289-294.
- Swanson KL, Woolley DE (1982) Comparison of the neurotoxic effects of chlordecone and dieldrin in the rat. *Neurotoxicology*. 3(2):81-102.
- Swartz, WJ; Mall, GM. (1989) Chlordecone-induced follicular toxicity in mouse ovaries. *Reprod Toxicol* 3:203-206.
- Swartz, WJ; Eroschenko, VP; Schutzmann, RL. (1988) Ovulatory response of chlordecone (Kepone)-exposed mice to exogenous gonadotropins. *Toxicology* 51(2-3):147-153.
- Taylor, JR. (1982) Neurological manifestations in humans exposed to chlordecone and follow-up results. *Neurotoxicology* 3(2):9-16.
- Taylor, JR. (1985) Neurological manifestations in humans exposed to chlordecone: follow-up results. *Neurotoxicology* 6(1):231-236.
- Taylor, JR; Selhorst, JB; Houff, SA; et al. (1978) Chlordecone intoxication in man: I. Clinical observations. *Neurology* 28:626-630.
- Teo, S; Vore, M. (1991) Mirex inhibits bile acid secretory function in vivo and in the isolated perfused rat liver. *Toxicol Appl Pharmacol* 109(1):161-170.
- Trosko, JE; Jone, C; Chang, CC. (1983) The role of tumor promoters on phenotypic alterations affecting intercellular communication and tumorigenesis. *Ann NY Acad Sci*. 407:316-327.
- Tsushimoto, G; Trosko, JE; Chang, CC; et al. (1982) Inhibition of intercellular communication by chlordecone (Kepone) and mirex in Chinese hamster v79 cells in vitro. *Toxicol Appl Pharmacol* 64:550-556.
- Ulland, BM; Page, NP; Squire, RL; et al. (1977) A carcinogenicity assay of mirex in Charles River CD rats. *J Natl Cancer Inst* 58:133-140.
- Uphouse, L. (1985) Effects of chlordecone on neuroendocrine function of female rats. *Neurotoxicology* 6(1):191-210.
- Uphouse, L; Eckols, K; Sierra, V; et al. (1986) Failure of chlordecone (Kepone) to induce behavioral estrus in adult ovariectomized rats. *Neurotoxicology* 7(1):127-142.

U.S. DHHS (Department of Health and Human Services). (2006) Glomerular diseases. National Kidney and Urologic Diseases Information Clearinghouse, Bethesda, MD; NIH Publication No. 07-4358. Available online at <http://kidney.niddk.nih.gov/kudiseases/pubs/glomerular/> (accessed June 25, 2009).

U.S. EPA (Environmental Protection Agency). (1986a) Guidelines for the health risk assessment of chemical mixtures. Federal Register 51(185):34014-34025. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).

U.S. EPA (Environmental Protection Agency). (1986b) Guidelines for mutagenicity risk assessment. Federal Register 51(185):34006-34012. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).

U.S. EPA. (Environmental Protection Agency). (1986c) Final report on the evaluation of four toxic chemicals in an in vivo/in vitro toxicological screen: acrylamide, chlordecone, cyclophosphamide, and diethylstilbestrol. EPA/600/1-86/002.

U.S. EPA. (Environmental Protection Agency). (1988) Recommendations for and documentation of biological values for use in risk assessment. Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Cincinnati, OH; EPA/600/6-87/008. Available from the National Technical Information Service, Springfield, VA; PB88-179874, and online at <http://cfpub.epa.gov/ncea/cfm/recorddisplay.cfm?deid=34855> (accesses June 25, 2009).

U.S. EPA (Environmental Protection Agency). (1991) Guidelines for developmental toxicity risk assessment. Federal Register 56(234):63798-63826. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).

U.S. EPA (Environmental Protection Agency). (1992) Draft report: a cross-species scaling factor for carcinogen risk assessment based on equivalence of mg/kg³/4/day. Federal Register 57(109):24152-24173.

U.S. EPA. (Environmental Protection Agency). (1994a) Interim policy for particle size and limit concentration issues in inhalation toxicity: notice of availability. Federal Register 59(206):53799. Available online at <http://www.epa.gov/EPA-PEST/1994/October/Day-26/pr-11.html> (accessed June 25, 2009).

U.S. EPA. (Environmental Protection Agency). (1994b) Methods for derivation of inhalation reference concentrations and application of inhalation dosimetry. Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Cincinnati, OH; EPA/600/8-90/066F. Available from the National Technical Information Service, Springfield, VA, PB2000-500023, and online at <http://cfpub.epa.gov/ncea/raf/recordisplay.cfm?deid=71993> (accessed June 25, 2009).

U.S. EPA. (Environmental Protection Agency). (1995) Use of the benchmark dose approach in health risk assessment. Risk Assessment Forum, Washington, DC; EPA/630/R-94/007. Available from the National Technical Information Service, Springfield, VA, PB95-213765, and online at http://cfpub.epa.gov/ncea/raf/raf_pubtitles.cfm?detype=document&excCol=archive (accessed June 25, 2009).

U.S. EPA (Environmental Protection Agency). (1996) Guidelines for reproductive toxicity risk assessment. Federal Register 61(212):56274-56322. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).

U.S. EPA. (Environmental Protection Agency). (1998) Guidelines for neurotoxicity risk assessment. Federal Register 63(93):26926-26954. Available online at <http://www.epa.gov/ncea/raf/rafguid.htm> (accessed January 15, 2009).

U.S. EPA. (Environmental Protection Agency). (2000a) Science policy council handbook: risk characterization. Office of Science Policy, Office of Research and Development, Washington, DC. EPA/100-B-00-002. Available from: <<http://www.epa.gov/iris/backgr-d.htm>>.

U.S. EPA (Environmental Protection Agency). (2000b) Benchmark dose technical guidance document. External review draft. Risk Assessment Forum, Washington, DC; EPA/630/R-00/001. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).

- U.S. EPA (Environmental Protection Agency). (2000c) Supplementary guidance for conducting health risk assessment of chemical mixtures. Risk Assessment Forum, Washington, DC; EPA/630/R-00/002. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).
- U.S. EPA (Environmental Protection Agency). (2000d) Status of chemicals in special review. EPA 738-R-00-0001
- U.S. EPA. (Environmental Protection Agency). (2002) A review of the reference dose concentration and reference concentration processes. Risk Assessment Forum, Washington, DC; EPA/630/P-02/002F. Available online at http://cfpub.epa.gov/ncea/raf/raf_pubtitles.cfm?detype=document&excCol=archive (accessed June 25, 2009).
- U.S. EPA (Environmental Protection Agency). (2005a) Guidelines for carcinogen risk assessment. Risk Assessment Forum, Washington, DC; EPA/630/P-03/001B. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).
- U.S. EPA (Environmental Protection Agency). (2005b) Supplemental guidance for assessing susceptibility from early-life exposure to carcinogens. Risk Assessment Forum, Washington, DC; EPA/630/R-03/003F. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).
- U.S. EPA (Environmental Protection Agency). (2006a) Science policy council handbook: peer review. Third edition. Office of Science Policy, Office of Research and Development, Washington, DC; EPA/100/B-06/002. Available online at <http://www.epa.gov/iris/backgr-d.htm> (accessed January 15, 2009).
- U.S. EPA. (2006b) A Framework for Assessing Health Risk of Environmental Exposures to Children. National Center for Environmental Assessment, Washington, DC, EPA/600/R-05/093F. Available from: <http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=158363>.
- Vaccari, A; Saba, P. (1995) The tyramine-labelled vesicular transporter for dopamine: a putative target of pesticides and neurotoxins. *Euro J Pharmacol* 292:309–314.
- Vig, PJS; Mehrotra, BD; Pennington, A; et al. (1989) Chlordecone interaction of calmodulin binding with phosphodiesterase. *FASEB J* 3(4):A1037.
- Wang, TP; Ho, IK; Mehendale, HM. (1981) Correlation between neurotoxicity and chlordecone (Kepone) levels in brain and plasma in the mouse. *Neurotoxicology* 2(2):373–381.
- Wang, F; Roberts, SM; Butfiloski, EJ; et al. (2007). Acceleration of autoimmunity by organochlorine pesticides: a comparison of splenic B-cell effects of chlordecone and estradiol in (NZB × NZW)F1 mice. *Toxicol Sci* 99:141–152.
- Wang, F; Sobel, ES; Butfiloski, EJ; et al. (2008) Comparison of chlordecone and estradiol effects on splenic T-cells in (NZB × NZW)F1 mice. *Toxicol Lett.* 183(1–3):1–9.
- WHO (World Health Organization). (1984) Chlordecone. Environmental health criteria. Vol. 43. International Programme on Chemical Safety, Geneva, Switzerland. Available online at <http://www.inchem.org/documents/ehc/ehc/EHC43.htm> (accessed June 25, 2009).
- Wiener, M; Pittman, KA; Stein, V. (1976) Mirex kinetics in the rhesus monkey. I. Disposition and excretion. *Drug Metab Dispos* 4(3):281–287.
- Williams, GM. (1980) Classification of genotoxic and epigenetic hepatocarcinogens using liver culture assays. *Ann NY Acad Sci* 349:273–282.
- Williams, GM. (1983) Epigenetic effects of liver tumor promoters and implications for health effects. *Environ Health Perspect* 50:177–183.
- Williams, GM; Numoto, S. (1984) Promotion of mouse liver neoplasms by the organochlorine pesticides chlordane and heptachlor in comparison to dichlorodiphenyltrichloroethane. *Carcinogenesis* 5:1689–1696.

Williams, J; Uphouse, L. (1991) Vaginal cyclicality, sexual receptivity, and eating behavior of the female rat following treatment with chlordecone. *Reprod Toxicol* 5(1):65–71.

Williams, J; Eckols, K; Uphouse, L. (1989) Estradiol and chlordecone interactions with the estradiol receptor. *Toxicol Appl Pharmacol* 98:413–421.

Williams, J; Montanez, S; Uphouse, L. (1992) Effects of chlordecone on food intake and body weight in the male rat. *Neurotoxicology* 13(2):453–462.

Yarbrough, JD; Chambers, JE; Grimley, JM; et al. (1981) Comparative study of 8-monohydromirex and mirex toxicity in male rats. *Toxicol Appl Pharmacol* 58(1):105–17.

Young, RA; Mehendale, HM. (1989) Carbon tetrachloride metabolism in partially hepatectomized and sham-operated rats pre-exposed to chlordecone (Kepone). *J Biochem Toxicol* 4(4):211–219.

APPENDIX A. SUMMARY OF EXTERNAL PEER REVIEW AND PUBLIC COMMENTS AND DISPOSITION

The Toxicological Review of Chlordecone has undergone a formal external peer review performed by scientists in accordance with U.S. EPA guidance on peer review (U.S. EPA, 2006a; 2000a). The external peer reviewers were tasked with providing written answers to general questions on the overall assessment and on chemical-specific questions in areas of scientific controversy or uncertainty. A summary of significant comments made by the external reviewers and U.S. EPA's responses to these comments arranged by charge question follow. In many cases the comments of the individual reviewers have been synthesized and paraphrased in development of Appendix A. U.S. EPA also received scientific comments from the public. These comments and U.S. EPA's responses are included in a separate section of this appendix.

On April 10, 2008, U.S. EPA introduced revisions to the IRIS process for developing chemical assessments. As part of the revised process, the disposition of peer reviewer and public comments, as found in this Appendix, and the revised IRIS Toxicological Review was provided to the external peer review panel members for their comment on May 4, 2009. No additional substantive comments were received as part of this second review.

EXTERNAL PEER REVIEW PANEL COMMENTS

The reviewers made several editorial suggestions to clarify specific portions of the text. These changes were incorporated in the document as appropriate and are not discussed further.

(A) General Comments

1. Is the Toxicological Review logical, clear, and concise? Has U.S. EPA accurately, clearly, and objectively represented and synthesized the scientific evidence for noncancer and cancer hazard?

Comments: All reviewers found the document to be generally logical and clear. One reviewer considered the Toxicological Review to be repetitive due to the format of the document. Two of the reviewers offered specific suggestions to improve the clarity of the document.

Response: U.S. EPA has reviewed and modified the Toxicological Review to address reviewer concerns about the repetitious nature of the document and to improve clarity.

2. Please identify any additional studies that should be considered in the assessment of the noncancer and cancer health effects of chlordecone.

Comments: Several reviewers identified the following additional literature:

Benachour, N; Moslemi, S; Sipahutar, H; et al. (2007) Cytotoxic effects and aromatase inhibition by xenobiotic endocrine disrupters alone and in combination. *Toxicol Appl Pharmacol* 222:129–140.

Hodges, LC; Bergerson, JS; Hunter, DS; et al. (2000). Estrogenic effects of organochlorine pesticides on uterine leiomyoma cells in vitro. *Toxicol Sci* 54(2):355–364.

Ray, S; Xu, F; Li, P; et al. (2007) Increased level of cellular Bip critically determines estrogenic potency for a xenoestrogen Kepone in the mouse uterus. *Endocrinology* 148(10):4774–4785.

Wang, F; Roberts, S; Butfiloski, EJ; et al. (2007) Diminished prolactin from chlordecone treatment in ovariectomized (NZB × NZW)F₁ mice. *Int Immunopharmacol* 7:1808–1812.

Wu, F; Safe, S. (2007) Differential activation of wild-type estrogen receptor alpha and C-terminal deletion mutants by estrogens, antiestrogens and xenoestrogens in breast cancer cells. *J Steroid Biochem Mol Biol* 103(1):1–9.

One reviewer also suggested that reviews on the carcinogenicity of chlordecone by authoritative bodies be considered (such as U.S. Department of Health and Human Services, International Agency for Research on Cancer, National Institute of Occupational Safety and Health, etc).

Response: Additional studies were considered and added where relevant. Specifically, references to Wang et al., 2007 were added to section 4.4.4 (Endocrine Disruption Studies) and section 4.4.5 (Immunological Studies), and Hodges et al., 2000 was added to section 4.4.4. Other in vitro studies suggested above were not considered to provide critical information for the health assessment of chlordecone and were not included. Reviews of carcinogenicity of chlordecone by other agencies or authoritative bodies were not included in the Toxicological Review as U.S. EPA's weight of evidence (WOE) for carcinogenicity is determined independently and is based on the approach presented in the U.S. EPA (2005a) *Guidelines for Carcinogen Risk Assessment*.

3. Please discuss research that you think would be likely to reduce uncertainty in the future assessments of chlordecone.

Comments: All reviewers commented that the addition of a multigenerational study would likely reduce uncertainty in future assessments of chlordecone. Other types of studies identified by multiple reviewers included immunological, neurodevelopmental, toxicokinetic, and mode of action studies for cancer and noncancer effects. One reviewer also suggested that a follow up study of chlordecone-exposed workers would contribute to future assessments of chlordecone.

Response: U.S. EPA agrees that the above research recommendations would improve future hazard identifications of chlordecone.

4. Please comment on the identification and characterization of sources of uncertainty in sections 5 and 6 of the assessment document. Please comment on whether the key sources of uncertainty have been adequately discussed. Have the choices and assumptions made in the discussion of uncertainty been transparently and objectively described? Has the impact of the uncertainty on the assessment been transparently and objectively described?

Comments: Four reviewers found the identification and characterization of sources of uncertainty to be reasonable. However, some of the reviewers identified additional areas of uncertainty that they believed were not addressed or should be discussed in more detail in Sections 5 and 6 of the Toxicological Review. These areas included:

- lack of mechanistic data and NOAELs for many observed effects
- potential early life vulnerabilities
- uncertainties between human and animal pharmacokinetics
- ability of existing studies to describe potential developmental toxicity
- wide range of blood levels associated with effects in exposed workers
- potential additive or synergistic effects of low doses of chlordecone with other similarly acting chemicals
- lack of establishment of NOAELs for female reproductive effects
- impact of the choice of BMD model chosen.

Response: Additional effort has been made to expand discussions of key areas of uncertainty in the Toxicological Review. Several of the uncertainties listed above have been addressed in section 6.2.1. of the document including the impact of the BMD model chosen and uncertainties between animal and human pharmacokinetics. A paucity of additional data to inform these uncertainties limits their further discussion in the document. Uncertainties regarding early life vulnerabilities were noted in section 4.8.1 and the lack of NOAELs for female reproductive effects was noted in section 6.2.1.

The uncertainty identified by a reviewer regarding potential additive effects of low doses of chlordecone with similarly acting chemicals is acknowledged as biologically plausible, however data on the mode(s) of action for cancer and noncancer effects of chlordecone are limited, as are data regarding the interactions of chlordecone with other chemicals which may share similar mechanisms. Therefore, further discussion of this point is considered speculative in light of the limited database for chlordecone.

Information to clarify the uncertainties regarding the wide range of blood levels in humans identified as suffering from chlordecone related symptoms (reported in Cannon et al., 1978) was added to section 3.1 and 6.2.1. Specifically, this large range of chlordecone blood levels may be partially explained by the authors' inclusive case definition as any worker with self-reported nervousness with or without objective neurological abnormalities (tremulousness, gait or motor abnormalities) upon examination. Workers with subjective symptoms alone represented 36% of identified cases. Chlordecone blood levels of the subset of workers with clinically confirmed neurological symptoms were not reported. Because of these uncertainties in measured blood levels, these data were not further used quantitatively.

(B) Oral reference dose (RfD) for Chlordecone

1. A chronic RfD for chlordecone has been derived from the 2-year dietary study (Larson et al., 1979a) in rats. Please comment on whether the selection of this study as the principal study has been scientifically justified. Has this study been transparently and objectively described in the document? Please identify and provide the rationale for any other studies that should be selected as the principal study.

Comments: Four reviewers agreed with the selection of Larson et al. (1979a) as the principal study. One reviewer suggested that more consideration should be given to the studies by Squibb and Tilson (1982) and Sobel et al. (2005, 2006) as potential principal studies for the derivation of the RfD. Two reviewers also commented that Linder et al. (1983) should be further considered as well.

Response: The sperm parameters reported in Linder et al. (1983) were examined for suitability for BMD modeling, and decreased sperm content was modeled and presented in Appendix B. Additionally, the possible POD from this endpoint was added to Figure 5-1 and Table 5-2. Endpoints observed in this study were considered for the derivation of the RfD, but these effects were observed at doses higher than the kidney toxicity demonstrated by Larson et al. (1979a). Thus, the sperm effects were not considered to be the most sensitive endpoint.

Studies by Sobel et al. were not considered for the development of the RfD due to the s.c. dosing regimen utilized.

The developmental study by Squibb and Tilson (1982) was reassessed and given greater consideration in the document. Body weights recorded at day 100 were statistically significantly decreased (19–27%) in both sexes of offspring at maternal doses of 0.6 mg/kg-day. These decreases in body weight were not accompanied by any visible signs of toxicity in any of the treatment groups. Recorded body weights at all other time points (PND 1, 7, 14, and 30) were no different from controls. Elimination studies of chlordecone indicate that pups would be expected

to be exposed to higher concentrations of chlordecone during the first 2 weeks of life (Egle et al., 1978); however, no significant effects on body weight were observed at early postnatal time points. No clear dose-response relationship was demonstrated in this study for decreased pup body weight. The only neurobehavioral endpoint that was affected by chlordecone exposure was a significant increase in the time required to reorient to a vertical position in an assay for negative geotaxis in male offspring exposed to 6 ppm at 100 days of age. The effect was not seen at 30 days in males and was not seen at either time point in female offspring. Motor activity induced by a dopamine receptor agonist was significantly increased in male offspring at 114 days of age in the high dose group 30 minutes after dosing and in both dose groups 60 minutes after dosing. In the absence of any clear neurological or behavioral response, the biological significance of the alteration in dopaminergic function associated with chlordecone exposure is uncertain. A LOAEL was determined for this review based on decreased body weight of female offspring at 100 days following a dietary maternal dose of 0.1 mg/kg-day chlordecone. In consideration of the uncertainties regarding the finding of decreased body weight in adult female offspring gestationally and lactationally exposed to chlordecone, including the latency, isolation, and lack of dose response of this finding coupled with the consideration that the LOAEL for this effect occurs in the range of the BMDL₁₀ for the dose-related increased incidence of kidney lesions in female rats, this finding was not considered the most appropriate effect on which to base the derivation of the RfD. A freestanding LOAEL for this effect was added as a possible POD to Figure 5-1 and Table 5-2.

Additionally, the dose conversion from 1 and 6 ppm into mg/kg-day was corrected based on current U.S. EPA dose conversion practices, which resulted in a slight change in mg/kg-day doses (0.07 mg/kg-day was corrected to 0.1 mg/kg-day and the high dose of 0.4 mg/kg-day was corrected to 0.6 mg/kg-day).

Comment: One reviewer pointed out that the study by Larson et al. (1979a), which initially comprised dose groups of 40 animals/sex, had greatly diminished numbers of animals available for examination at the chronic time point, with only four animals/group examined in the 1.6 mg/kg-day group at 1 year. It was requested that descriptions of this study design be revised to reflect the number of animals evaluated at study termination.

Response: The number of animals in this study available for histological examination at study termination appear to be limited by periodic serial sacrifices during the study and decreased survival of animals in the high dose groups. Clarifications of these limitations have been added to the study description in Sections 4.2 and 5.1.1.

2. Kidney (glomerular) lesions, liver lesions, and reproductive effects are all sensitive effects of chlordecone exposure. Glomerular lesions in the kidney was selected as the most appropriate

critical effect. Please comment on whether the selection of glomerular lesions as the critical effect instead of reproductive endpoints (such as testicular lesions) has been scientifically justified. Is this choice transparently and objectively described in the document? Please provide detailed explanation. Please identify and provide the rationale for any other endpoints that should be considered in the selection of the critical effect.

Comment: Most of the reviewers supported the selection of kidney lesions as the critical effect with the caveat that effects observed in the studies by Linder et al. (1983) Sobel et al. (2005, 2006), and Squibb and Tilson (1982) should be given greater consideration.

Response: See response to Charge Question 1.

Comment: One reviewer expressed concern with the use of the 15-day study by Chetty et al. (1993) as support for glomerulosclerosis as the critical effect due to the observation of statistically significant changes in liver enzyme changes (i.e., SGPT enzymes changes) at doses lower (>1.0 mg/kg) than those at which kidney parameters were significantly altered (>4.9 mg/kg-day).

This reviewer also commented that in the Larson et al. (1979a) study, the high background incidence of kidney lesions in control male rats (55%) contributes to uncertainty regarding the selection of kidney effects in female rats (which had a lower background incidence of lesions, 12%) as the critical effect for the basis of the RfD.

Response: The short-term study of chlordecone exposure by Chetty et al. (1993) was discussed to demonstrate additional support for the observed dose-related kidney effects, specifically glomerular in nature, following chlordecone exposure in Larson et al. (1979a). The changes in both the liver and kidney parameters observed by Chetty et al. (1993) occurred at higher doses than the kidney effects observed in Larson et al. (1979a). U.S. EPA recognizes that the alterations in serum liver enzyme levels occur at a lower dose than the significantly elevated serum indicators of kidney damage in the Chetty et al. (1993) study. However, Chetty et al. (1993) is of short duration and does not inform which of these effects would be expected to be most sensitive following low dose chronic exposure to chlordecone.

Evidence from the Larson et al. (1979a) study indicates kidney effects as the most sensitive effect observed in rats following chronic exposure of chlordecone. It is possible that the occurrence of the high background of age-related kidney lesions in the test species contributes to the observation of kidney lesions as the most sensitive dose-related effect following chlordecone exposure. However, age-related decline in the glomerular function of humans also occurs with the presence of androgens as a prominent risk factor (this may explain the higher background incidence of kidney lesions in male animals) (Baylis, 1994). In the

absence of data to indicate that the MOA of kidney lesions observed in rats would not be relevant to humans, the most sensitive observed effect in the chronic rat study by Larson et al. (1979a) was selected as the most appropriate critical effect. Additional information regarding the uncertainties of the high background incidence of kidney lesions in male rats has been added to section 6.2.1.

Comment: One reviewer was concerned that the decreased sperm parameters (sperm viability, motility, and epididymal reserves) reported in Linder et al. (1983) were also affected at the lowest dose level (0.26 mg/kg-day), which the text refers to as a NOAEL. The reviewer states that a larger number of rats may have shown the effect to be significant and the text should not dismiss this dose as a NOAEL.

Response: A statistically significant decrease in sperm parameters was observed at the level of the 0.86 mg/kg-day dose group. Therefore, taking into consideration the biological significance of these effects along and with the observed dose-response, the NOAEL assigned was 0.26 mg/kg-day. U.S. EPA agrees that the determination of NOAELs and LOAELs is highly dependent on study design and that the use of larger groups of animals would potentially result in larger power to detect effects at lower doses.

3. Some evidence exists to suggest that the mechanism of the critical effect selected for determination of the POD (i.e., glomerular lesions) may be mediated through an autoimmune mechanism. Please comment on whether the available immunotoxicity data support this proposed MOA. Is this proposed MOA scientifically justified and transparently described?

Comments: All of the reviewers agreed that the proposed MOA for the observed kidney lesions was plausible and that the discussion was reasonable and transparent.

Response: No response.

4. The chronic RfD has been derived utilizing BMD modeling to define the POD. All available models were fit to the data for the incidence of glomerulosclerosis in female rats. Please provide comments with regards to whether BMD modeling is the best approach for determining the POD. Has the BMD modeling been appropriately conducted and objectively and transparently described? Has the BMR selected for use in deriving the POD been scientifically justified? Is it transparently and objectively described? Please identify and provide rationale for any alternative approaches (including the selection of BMR, model, etc.) for the determination of the POD, and if such approaches are preferred to U.S. EPA's approach.

Comments: All reviewers agreed that BMD modeling was the best approach for determining the POD. Additionally, none of the reviewers disagreed with the BMR of 10%. One reviewer proposed averaging the BMDLs from the models with acceptable fit (p -value > 0.1) and very small differences in AIC values instead of choosing the model with the lowest AIC value. Another reviewer suggested increased rationale for model section.

Response: In concordance with U.S. EPA's *Benchmark Dose Guidance* (U.S. EPA, 2000c), models were assessed by a chi-square goodness-of-fit test. The model exhibiting adequate fit with the lowest AIC value, which provides a measure of the deviance of the model fit and allows for comparison across models for a particular endpoint, was selected in accordance with the guidance. Increased rationale for model section was included in Section 5.1.2 and Appendix B.

5. Please comment on the selection of the uncertainty factors applied to the POD for the derivation of the RfD. For instance, are they scientifically justified and transparently and objectively described in the document?

Comments: The reviewers generally agreed that the selection and justification for the uncertainty factors of 10 for the extrapolation from animals to humans and the consideration of variability between humans was reasonable. However, some of the reviewers suggested that the UF of 10 for interspecies extrapolation could be reduced using a cross species $BW^{3/4}$ dosimetric adjustment for differences between rat and human kinetics and an UF of 3 to account for pharmacodynamic differences.

Additionally, some of the reviewers commented on the application of uncertainty factors selected for potential PODs in Table 5-2 and Figure 5-1. Specifically, a 10X subchronic to chronic uncertainty factor for testicular atrophy was argued to be inappropriate as this endpoint was examined at a chronic duration in the same study and not detected.

Response: It is currently U.S. EPA's default policy to use an UF of 10 to take into consideration toxicokinetic and toxicodynamic differences between animal test species and humans in the extrapolation of an oral Reference Dose (RfD) in the absence of chemical-specific data or biologically based models to inform a human equivalent dose. It is current Agency practice to use $BW^{3/4}$ scaling in the derivation of oral slope factors (U.S. EPA, 2005a). In an effort to harmonize interspecies extrapolation in cancer and noncancer dose-response methodologies, $BW^{3/4}$ scaling is being evaluated, though it has not yet been adopted as default practice for dosimetric adjustments in the calculation of an oral RfD.

Table 5-2 and Figure 5-1 are for comparison purposes only, since these potential PODs across studies are not necessarily comparable, nor is the confidence the same in the data sets from which the PODs were derived. Nevertheless, the additional subchronic to chronic

uncertainty factor applied to POD for testicular lesions was reduced as recommended, as this endpoint was examined at a chronic duration in the same study and not detected.

6. An uncertainty factor was considered necessary to account for deficiencies in the chlordecone toxicity database (e.g., absence of standard two-generation reproduction studies and immunotoxicity studies). Please comment on whether the rationale and justification for the application of the database uncertainty factor has been scientifically justified and transparently described in the document. Please comment on whether the available immunotoxicity data for chlordecone indicate that additional immunological studies could result in a different POD.

Comments: Two of the reviewers supported a database uncertainty factor of 3 or less. Specifically, one reviewer did not believe that a two generation reproductive study would lead to effects that had not already been identified by the various reproductive or chronic studies already conducted or provide a lower POD. The other reviewer recommended that additional rationale for the selection of a database uncertainty factor of 3 should be included in the assessment. Three of the reviewers commented that the database UF should be increased to 10. Rationale for increasing the database UF presented by the panel included:

- Absence of an adequate cancer bioassay and lack of NOAELs for noncancer endpoints (e.g., female reproductive endpoints).
- Evidence of neurological and estrogenic activity of chlordecone.
- Presence of endpoints proposed to be lacking clear NOAELs (including renal immunotoxicity, spermatotoxicity, developmental neurotoxicity, and frank toxicity) and possible increased sensitivity of susceptible human populations (such as individuals with greater susceptibility to Lupus).

Additionally, three reviewers commented that they did not support the application of a database uncertainty factor based on the lack immunotoxicity data (the remaining two reviewers did not comment).

Response: The database uncertainty factor of 3 was retained as a default uncertainty factor of 10 was not considered necessary to account for deficiencies in the chlordecone toxicity database. The database includes several studies in laboratory animals, including chronic and subchronic dietary exposure studies and several reproductive and developmental studies, including one where developmental neurotoxicity was assessed. Though the chlordecone database does not have a standard multigenerational reproductive study, it does contain over ten oral repeat exposure studies assessing reproductive and developmental toxicity including several single generation reproductive toxicity studies and three adequately designed developmental studies in rats and mice (Good et al., 1965; Huber et al., 1965; Cannon and Kimbrough, 1979; Linder et al.,

1983; Chernoff and Rogers, 1976; Squibb and Tilson). Additionally, a few reproductive studies exist in the chlordecone database with a multigenerational component (Good et al., 1965; Gellert and Wilson 1979). However, further rationale was provided in Section 5.1.3 to support the 3-fold database uncertainty factor.

Several reviewers recommended increasing the database uncertainty factor based on the points described in the comment above. Specifically, one reviewer recommended that uncertainty related to potentially susceptible populations (i.e., individuals with Lupus) should be considered. While this group is recognized as a potentially susceptible population, the intraspecies uncertainty factor (UF_H) was applied to account for human variability and susceptibility in response to chlordecone. Therefore, an increase in the database uncertainty factor to account for this population is inappropriate.

Another reviewer stated the database UF should be increased to account for the lack of an adequate cancer bioassay given the available data demonstrating chlordecone-induced hepatocellular carcinoma. This uncertainty is not considered as part of the RfD derivation, but is taken into account in the characterization of the weight of the evidence for carcinogenicity. Therefore, an increase in the database uncertainty factor to account for this is inappropriate.

One reviewer noted that data suggesting estrogenic effects, including alterations of the female estrous cycle, following exposure to chlordecone justified an increase in the database uncertainty factor. Studies in mice by Swartz et al (1988) and Huber (1965) reported animals in a state of persistent vaginal estrus (PVE) at doses ≥ 2 mg/kg-day (greater than the POD of 0.08 mg/kg-day). Alterations of the estrous cycle may indicate disruption of ovulation and the subsequent reduction of fertility, though subtle alterations of cyclicity can occur at doses below those that alter fertility (U.S. EPA 1996). Studies by Swartz et al. (1988) and Huber (1965) showed reductions in fertility at doses ≥ 5.6 mg/kg-day. Therefore, uncertainty exists as to whether alterations in estrous cycling would be expected at doses around the point of departure (0.08 mg/kg-day), though it is unlikely that detectable reduction of fertility would be observed. A guideline multigeneration reproduction study would evaluate fertility and male and female reproductive endpoints, including estrous cycle effects which are currently not well characterized. Therefore, EPA concluded that the 3-fold database UF applied for the lack of a two-generation reproductive study accounts for the possibility of detecting estrogenic effects at levels below the current point of departure.

Several reviewers proposed that the presence of endpoints lacking clear NOAELs (including neurotoxicity, spermatotoxicity, estrogenicity, developmental neurotoxicity, renal immunotoxicity, frank toxicity, and female reproductive toxicity) in the chlordecone database justified an increased database uncertainty factor. The existing NOAELs and LOAELs for these effects have been investigated, but occur at doses above those inducing the critical effect of increased glomerular lesions in female rats (see Table 4-18). One reviewer believed the NOAEL for renal immunotoxicity was undefined in studies conducted in ovariectomized mice (Sobel et

al., 2006). This study reported a NOAEL and LOAEL for decreased latency of renal effects associated with the subcutaneous implantation of 0.1 mg and 0.5 mg tablets, respectively; however, due to the use of subcutaneous dosing, this study is not appropriate for the derivation of an oral RfD.

Additionally, several of the reviewers disagreed that available immunotoxicity data for chlordecone indicates that additional immunological studies would likely result in a lower POD. Therefore, text in support of the database uncertainty factor was revised in the Toxicological Review to indicate that the database uncertainty factor of 3 was applied to account for the lack of a standard multigeneration reproduction study, not immunotoxicity data deficiencies.

(C) Carcinogenicity of Chlordecone

1. Under the EPA's 2005 *Guidelines for Carcinogen Risk Assessment* (<http://cfpub.epa.gov/ncea/raf/recordisplay.cfm?deid=116283>), there is *suggestive evidence of the human carcinogenic potential* of chlordecone. This characterization lies at the high end of the continuum for this weight of evidence descriptor. Please comment on the scientific justification for the cancer weight of evidence characterization. Has the scientific justification for the weight of evidence characterization been sufficiently, transparently, and objectively described? A quantitative cancer assessment has not been derived for chlordecone. Do the data support an estimation of a cancer slope factor for chlordecone? Please comment on the scientific justification for not deriving a quantitative cancer assessment considering the uncertainty in the data and the suggestive nature of the weight of evidence of carcinogenic potential.

Comments: Two reviewers agreed with the conclusion of suggestive evidence of carcinogenicity and also concurred with the decision not to quantify cancer risk due to the limitations of the only cancer bioassay and the lack of evidence for genotoxicity and potential for promotion as the mode of action. Three of the reviewers believed that greater than suggestive evidence for carcinogenic potential exists for chlordecone based on the findings of liver tumors in two sexes of two species. One of these three reviewers, who believed chlordecone is likely to be carcinogenic to humans, supported the decision not to quantify cancer risk citing the bioassay design and conduct limitations, the lack of genotoxicity evidence, and evidence for a promotional mode of action. However, the two other reviewers indicated that the decision not to quantify cancer risk was not well supported and the cited irregularities in the dose-response data and early mortality reported in the study could be accounted for in the quantitative cancer assessment.

Response: U.S. EPA agrees with several of the reviewers that based upon the 2005 U.S. EPA *Guidelines for Carcinogen Risk Assessment*, chlordecone is “likely to be carcinogenic to humans” based on the statistically significant increased incidence of liver tumors in two sexes of two species. In light of this strengthened descriptor for the weight of evidence of carcinogenicity, the U.S. EPA re-evaluated methods to quantify this risk.

Inadequate data exist for chlordecone to determine a MOA for the liver carcinomas observed in both sexes of rats and mice. Thus, in the absence of a MOA for the observed liver tumors, a low dose linear extrapolation was performed to quantify cancer risk.

U.S. EPA took into consideration the high toxicity resulting in early deaths and used time to tumor type modeling of the data. Upon time-to-tumor modeling of all datasets from the NCI (1976a, b) studies, it was determined that this method could be used to estimate an oral slope factor for all datasets but one (the high dose male rats). Details of this analysis are presented in section 5.3 and in Appendix C.

U.S. EPA acknowledges that uncertainty exists in the data and the methods used to derive an oral slope factor for chlordecone. Section 5.3.5 was added to qualitatively discuss these uncertainties in the quantitative cancer assessment.

Additional Comments

Comment: One reviewer requested that it would be beneficial to compare and discuss the blood levels associated with effects in humans with those calculated for BMDs and LOAELs from the animal studies, using pharmacokinetic approaches.

Response: Currently, no published PBTK models exist which predict blood levels of chlordecone following oral administration of chlordecone; therefore, pharmacokinetic approaches are not readily available to compare BMDs and LOAELs from animal studies to blood levels observed in humans occupationally exposed to chlordecone.

PUBLIC COMMENTS

Comment: One public commenter pointed out that residual chlordecone contamination of fish and shellfish is an ongoing issue in the James River area of Virginia. This commenter pointed out that the “suggestive” potential for carcinogenicity of chlordecone, as presented in U.S. EPA’s Draft Toxicological Review, without the development of a quantitative value of cancer potency, hampers the ability of risk assessors to reach a conclusion regarding potential risks of residual chlordecone exposure and recommended the calculation of a cancer potency factor or an upper range of a plausible cancer potency factor for chlordecone.

Response: In response to the comments of the external peer reviewers, U.S. EPA has re-evaluated the WOE for carcinogenicity of chlordane and the feasibility of quantifying an oral slope factor from the data while considering and acknowledging the uncertainties inherent in the data.

APPENDIX B. BENCHMARK DOSE CALCULATIONS FOR THE RfD

Kidney Lesions (Glomerulosclerosis) in Female Rats Exposed to Chlordecone in the Diet for 1–2 years

The Larson et al. (1979a) study did not include statistics for renal lesions as described in Section 4.2.2. Statistical analysis (performed for this review) of the frequency of renal lesions in each dose by sex (Fisher's exact test) revealed that the incidence of glomerulosclerosis (grades 1, 2, or 3 combined) in almost all of the exposure groups of female rats was statistically different from control. Additionally, a significant dose-response trend was seen by the Cochran-Armitage test. All available dichotomous models in the U.S. EPA BMDS version 1.3.2 were fit to the quantal incidence data (Table B-1) for histopathologic glomerulosclerosis in female Wistar rats from a 2-year dietary study (Larson et al., 1979a). To provide potential points of departure for RfD derivation, BMR levels of 10% extra risk for quantal incidence data were selected in the absence of biological information that would warrant a different choice and under the assumption that it represents a minimal biologically significant change (U.S. EPA, 2000c). The results of statistical analysis and BMD modeling for each sex are described below.

Table B-1. Incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in female Wistar rats following administration of chlordecone in the diet for 2 years

| Gender | Dose (mg/kg-day) | | | | |
|---------------------|------------------|-------------|-------------------------|-------------------------|------------------------|
| | 0 | 0.06 | 0.3 | 0.5 | 1.6 |
| Male | 12/22 | 3/11 | 4/6 | 6/9 | 3/4 |
| Female ^a | 4/34 | 2/13 | 8/17^b | 8/12^b | 3/4^b |

^aStatistically significant trend for increased incidence by Cochran-Armitage test ($p < 0.01$).

^bStatistically significantly different from controls according to Fisher's exact test ($p < 0.05$) performed for this review.

Source: Larson et al. (1979a).

As shown in Table B-1, the frequency of renal lesions (glomerulosclerosis) in female rats was statistically significantly different from the incidence among control rats at doses of ≥ 0.3 mg/kg-day. Most dichotomous models provided adequate fit to the female rat incidence data, based on the summary results reported in the BMDS output and a more detailed examination of the graphs and chi-square goodness-of-fit statistics (summarized in Table B-2 and Figure B-1).

Two of the seven dichotomous models in BMDS (the logistic and probit models) exhibited significant lack of fit. As shown in Table B-2, the remaining five models in BMDS

provided sufficient fit to the data as assessed through χ^2 *p*-value. Of these five models, the multistage, Weibull, and gamma models yielded identical fits, essentially reducing the number of adequately fitting models to three. BMDL₁₀ estimates from the models were within a factor of three showing no appreciable model dependence. The model with the lowest AIC value (i.e., a measure of the deviance of the model fit that allows for comparison across models for a particular endpoint) was selected as the best-fit model (U.S. EPA, 2000c). The log-probit model yielded the lowest AIC (i.e., 84.3) and resulted in BMD₁₀ and BMDL₁₀ estimates of 0.12 and 0.08 mg/kg-day, respectively, associated with a 10% extra risk for glomerulosclerosis (Table B-2). BMD output from the log-probit model is included below.

Table B-2. BMD modeling results for the incidence of histopathologic renal lesions (glomerulosclerosis) in female Wistar rats, following administration of chlordecone in the diet for 2 years

| Model | BMD ₁₀ | BMDL ₁₀ | χ^2 <i>p</i> -value | AIC |
|----------------------------|-------------------|--------------------|--------------------------|------|
| Log-probit | 0.116 | 0.076 | 0.62 | 84.3 |
| Multistage, Weibull, Gamma | 0.071 | 0.045 | 0.56 | 84.7 |
| Log-logistic | 0.067 | 0.026 | 0.72 | 85.7 |

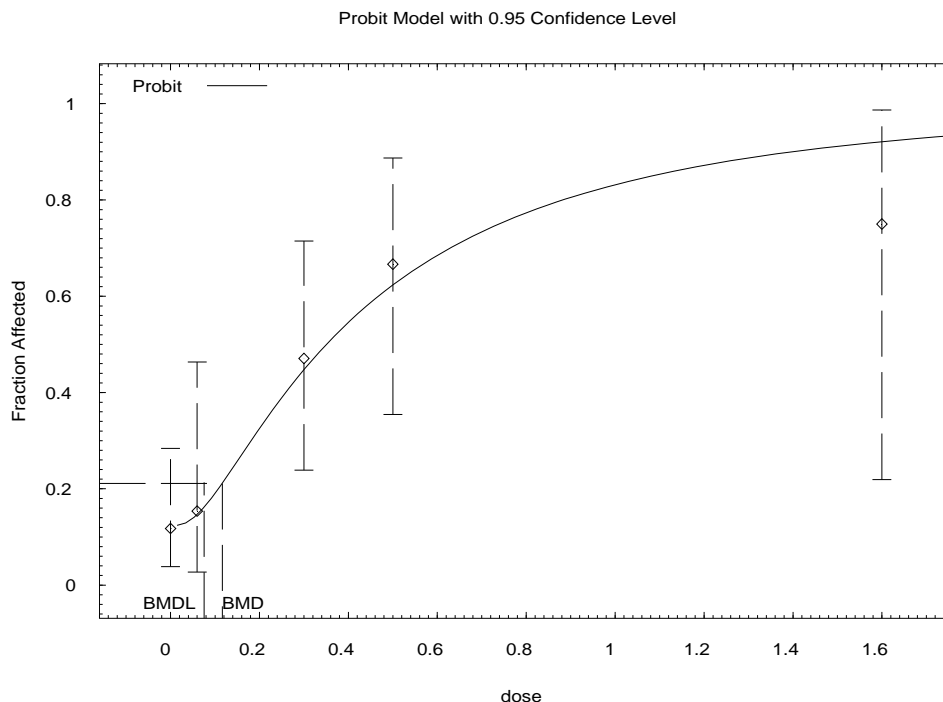


Figure B-1. Observed and predicted incidence of histopathologic renal lesions (glomerulosclerosis grades 1, 2, or 3 combined) in female Wistar rats following administration of chlordecone in the diet for 1–2 years. Log-Probit Model of U.S. EPA BMDS (Version 1.3.2).

The computer output from the log-Probit model of the glomerulosclerosis data follows:

Probit Model \$Revision: 2.1 \$ \$Date: 2000/02/26 03:38:53 \$
Input Data File: C:\BMDS\KIDNEY_LESIONS.(d)
Gnuplot Plotting File: C:\BMDS\KIDNEY_LESIONS.plt
Wed May 09 15:06:56 2007

BMDS MODEL RUN

The form of the probability function is:

$$P[\text{response}] = \text{Background} + (1 - \text{Background}) * \text{CumNorm}(\text{Intercept} + \text{Slope} * \text{Log}(\text{Dose})),$$

where CumNorm(.) is the cumulative normal distribution function

Dependent variable = COLUMN1
Independent variable = COLUMN3
Slope parameter is restricted as slope ≥ 1

Total number of observations = 5
Total number of records with missing values = 0
Maximum number of iterations = 250
Relative Function Convergence has been set to: 1e-008
Parameter Convergence has been set to: 1e-008

User has chosen the log transformed model

Default Initial (and Specified) Parameter Values
background = 0.117647
intercept = 0.723913
slope = 1

Asymptotic Correlation Matrix of Parameter Estimates

(*** The model parameter(s) -slope
have been estimated at a boundary point, or have been specified by the user,
and do not appear in the correlation matrix)

| | background | intercept |
|------------|------------|-----------|
| background | 1 | -0.36 |
| intercept | -0.36 | 1 |

Parameter Estimates

| Variable | Estimate | Std. Err. |
|------------|----------|-----------|
| background | 0.123642 | 0.0510126 |
| intercept | 0.869701 | 0.276028 |
| slope | 1 | NA |

NA - Indicates that this parameter has hit a bound implied by some inequality constraint and thus has no standard error.

Analysis of Deviance Table

| Model | Log(likelihood) | Deviance | Test DF | P-value |
|---------------|-----------------|----------|---------|-----------|
| Full model | -39.5379 | | | |
| Fitted model | -40.1501 | 1.22434 | 3 | 0.7472 |
| Reduced model | -49.6869 | 20.2979 | 4 | 0.0004361 |

AIC: 84.3002

Goodness of Fit

| Dose | Est._Prob. | Expected | Scaled | | Residual |
|--------|------------|----------|----------|------|----------|
| | | | Observed | Size | |
| 0.0000 | 0.1236 | 4.204 | 4 | 34 | -0.1062 |
| 0.0600 | 0.1464 | 1.903 | 2 | 13 | 0.07598 |
| 0.3000 | 0.4471 | 7.601 | 8 | 17 | 0.1948 |
| 0.5000 | 0.6232 | 7.479 | 8 | 12 | 0.3105 |
| 1.6000 | 0.9210 | 3.684 | 3 | 4 | -1.268 |

Chi-square = 1.76 DF = 3 P-value = 0.6241

Benchmark Dose Computation

Specified effect = 0.1

Risk Type = Extra risk

Confidence level = 0.95

BMD = 0.116338

BMDL = 0.0756267

Testicular Atrophy in Male Rats Receiving Chlordecone in the Diet for 3 months

The Larson et al. (1979a) study did not include statistics for the testicular atrophy observed in male rats (see Section 4.2.2). Statistical analysis (performed for this review) of the frequency of renal lesions in each dose by sex (Fisher's exact test) revealed that the incidence of testicular atrophy in some of the exposure groups of male rats was statistically different from control. Additionally, a significant dose response trend was seen by the Cochran-Armitage test. All available models in the U.S. EPA BMDS version 1.3.2 were fit to quantal incidence data (Table B-3) for testicular atrophy in male Wistar rats, following 3 months of dietary exposure (Larson et al., 1979a). To provide potential PODs for RfD derivation, BMR levels of 10% extra risk for quantal incidence data were selected. The results of statistical analysis and BMD modeling for each sex are described below.

As shown in Table B-3, the frequency of testicular atrophy in male rats was statistically different from the incidence among control rats at doses of ≥ 1.6 mg/kg-day. However, the highest dose groups of 3.9 and 7 mg/kg-day were not included in the dose response modeling as animals in these dose groups suffered from overt toxicity, leading to death of all animals in these groups by 6 months into the study. Testicular atrophy in the highest exposed rats may have resulted from frank toxic effects including decreased body weight gain.

All of the dichotomous models provided adequate fit to the testicular atrophy incidence data based on the summary results reported in the BMDS output and a more detailed examination of the graphs and chi-square goodness-of-fit statistics (summarized in Table B-4 and Figure B-2). As shown in Table B-4, the multistage model provided the best fit as indicated by the lowest AIC value (Figure B-2). This model predicted the BMD₁₀ associated with a 10% extra risk for testicular atrophy as 0.21 mg/kg-day (Table B-4). The BMDL₁₀, a potential POD for the reference dose (RfD), was 0.12 mg/kg-day (Table B-4).

Table B-3. Incidence of testicular atrophy in male rats receiving chlordecone in the diet for 3 months

| Dietary level (ppm) | 0 | 5 | 10 | 25 | 50 | 80 |
|--|----------|----------|-----------|------------------|------------------|------------------|
| Average dose ^a (mg/kg-day) | 0 | 0.3 | 0.5 | 1.6 | 3.9 | 7.0 |
| Incidence of testicular atrophy ^b | 1/10 | 0/5 | 1/5 | 4/5 ^c | 4/5 ^c | 5/5 ^c |

^aAverage doses to male rats, based on graphically depicted food consumption data presented by the authors.

^bStatistically significant trend for increased incidence by Cochran-Armitage test ($p < 0.01$).

^cStatistically significantly different from controls according to Fisher's exact test ($p < 0.05$) performed for this review.

Source: Larson et al. (1979a).

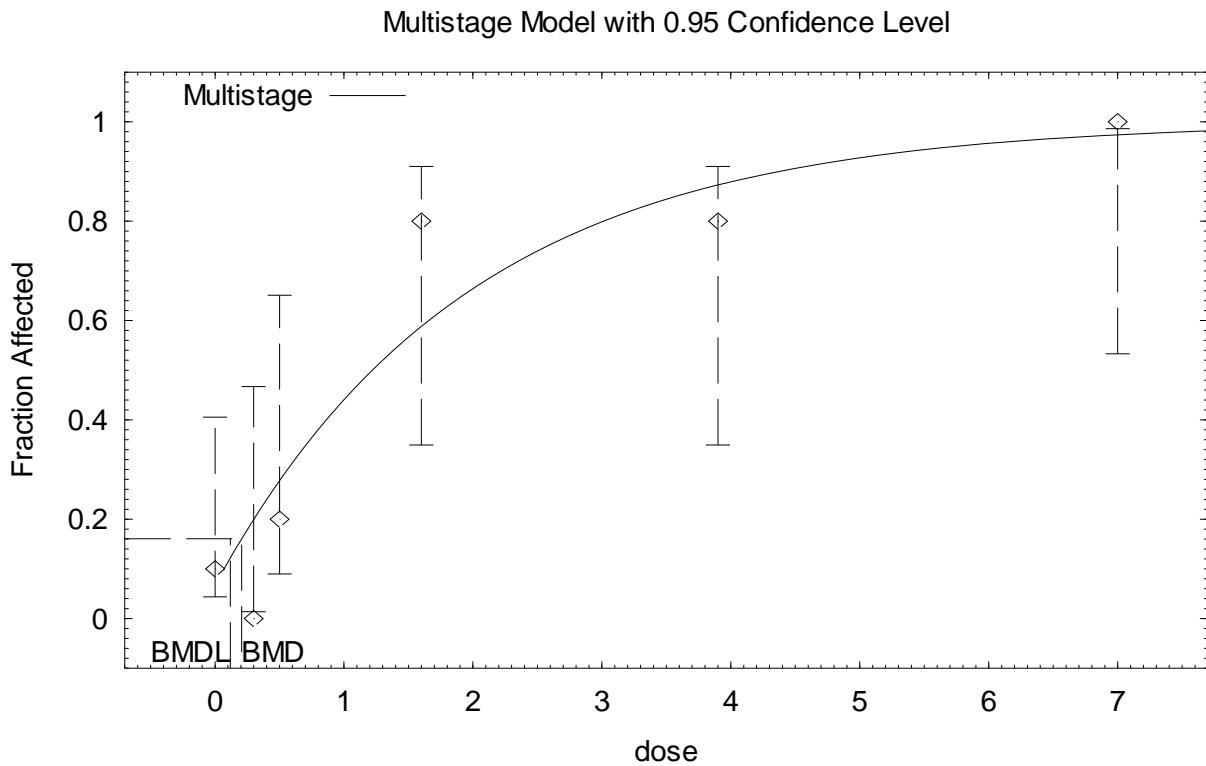
Table B-4. BMD modeling results for the incidence of testicular atrophy in male Wistar rats, following administration of chlordecone in the diet for 3 months

| Model | BMD ₁₀ | BMDL ₁₀ | χ^2 p-value | AIC |
|---|-------------------|--------------------|------------------|--------------|
| Gamma | 0.393 | 0.126 | 0.42 | 30.97 |
| Logistic | 0.560 | 0.323 | 0.35 | 30.52 |
| Log-logistic | 0.436 | 0.125 | 0.49 | 30.37 |
| Multistage (1^o)^a | 0.206 | 0.119 | 0.58 | 29.54 |
| Probit | 0.563 | 0.350 | 0.36 | 30.58 |
| Log-probit | 0.444 | 0.203 | 0.51 | 30.28 |
| Weibull | 0.338 | 0.123 | 0.41 | 31.15 |

^aForm of the multistage model:

$$P[\text{response}] = \text{background} + (1 - \text{background}) \times (1 - \text{EXP}(-\text{beta} \times \text{dose}^{-1}))$$

Where: background = 0.0672234; beta(1) = 0.510742.



11:39 05/09 2007

Figure B-2. Observed and predicted incidence of testicular atrophy in male Wistar rats, following administration of chlordecone in the diet for 3 months.

Multistage Model of U.S. EPA BMDS (Version 1.3.2).

The computer output from the Multistage model of the male testicular atrophy follows:

```
=====
Multistage Model. $Revision: 2.1 $ $Date: 2000/08/21 03:38:21 $
Input Data File: G:\KEPONE DOSE-RESPONSE
MODELING\MALE_RAT_TESTES_LARSON_1979.(d)
Gnuplot Plotting File: G:\KEPONE DOSE-RESPONSE
MODELING\MALE_RAT_TESTES_LARSON_1979.plt
Wed May 09 11:39:01 2007
=====
```

BMDS MODEL RUN

The form of the probability function is:

$$P[\text{response}] = \text{background} + (1 - \text{background}) * [1 - \text{EXP}(-\text{beta}1 * \text{dose}^1)]$$

The parameter betas are restricted to be positive

Dependent variable = Response
Independent variable = Dose

Total number of observations = 6
Total number of records with missing values = 0
Total number of parameters in model = 2
Total number of specified parameters = 0
Degree of polynomial = 1

Maximum number of iterations = 250
Relative Function Convergence has been set to: 1e-008
Parameter Convergence has been set to: 1e-008

Default Initial Parameter Values
Background = 0
Beta(1) = 1.27121e+019

Asymptotic Correlation Matrix of Parameter Estimates

Background Beta(1)

Background 1 -0.41
 Beta(1) -0.41 1

Parameter Estimates

| Variable | Estimate | Std. Err. |
|------------|-----------|-----------|
| Background | 0.0672234 | 0.22791 |
| Beta(1) | 0.510742 | 0.227823 |

Analysis of Deviance Table

| Model | Log(likelihood) | Deviance | Test DF | P-value |
|---------------|-----------------|----------|---------|---------|
| Full model | -10.7569 | | | |
| Fitted model | -12.7712 | 4.02865 | 4 | 0.4021 |
| Reduced model | -23.9018 | 26.2898 | 5 | <.0001 |

AIC: 29.5424

Goodness of Fit

| Dose | Est._Prob. | Expected | Observed | Size | Chi^2 Res. | |
|------|------------|----------|----------|------|------------|--------|
| i: 1 | 0.0000 | 0.0672 | 0.672 | 1 | 10 | 0.523 |
| i: 2 | 0.3000 | 0.1997 | 0.999 | 0 | 5 | -1.250 |
| i: 3 | 0.5000 | 0.2774 | 1.387 | 1 | 5 | -0.386 |
| i: 4 | 1.6000 | 0.5880 | 2.940 | 4 | 5 | 0.875 |
| i: 5 | 3.9000 | 0.8727 | 4.364 | 4 | 5 | -0.655 |
| i: 6 | 7.0000 | 0.9739 | 4.869 | 5 | 5 | 1.027 |

Chi-square = 2.87 DF = 4 P-value = 0.5800

Benchmark Dose Computation

Specified effect = 0.1

Risk Type = Extra risk

Confidence level = 0.95

BMD = 0.206289

BMDL = 0.118596

Liver Lesions (Fatty Changes and Hyperplasia) in Male and Female Rats Exposed to Chlordecone in the Diet for 1–2 years

The Larson et al. (1979a) study did not include statistics for liver lesions. Statistical analysis, performed for this review, of the frequency of liver lesions in each dose by sex (Fisher's exact test and Cochran-Armitage trend test) revealed that the incidence of liver lesions in some of the exposure groups was statistically different from controls. An examination of liver lesion incidence based on sex indicated (by Fisher's exact test) no significant differences; the incidence data for males and females were combined. The incidence data were used to fit the dichotomous models available in the U.S. EPA BMDS version 1.3.2. The frequency of liver lesions (fatty changes and hyperplasia) in both sexes combined was statistically different from control at 0.5 and 1.6 mg/kg-day (see Table B-5). In addition, the Cochran-Armitage trend test showed a statistically significant dose-response trend in the frequency of liver lesions (fatty changes and hyperplasia) for both sexes combined.

Table B-5. Incidence of histopathologic liver lesions (fatty changes and hyperplasia) in Wistar rats, following administration of chlordecone in the diet for 1–2 years

| Endpoint | Dose (mg/kg-day) | | | | |
|----------------------------|------------------|------|------|-------------------|------------------|
| | 0 | 0.06 | 0.3 | 0.5 | 1.6 |
| Liver lesions ^a | | | | | |
| Male rats | 1/22 | 1/11 | 2/6 | 2/9 | 3/4 ^b |
| Female rats | 2/34 | 1/13 | 2/17 | 4/12 ^b | 1/4 |
| Both | 3/56 | 2/24 | 4/23 | 6/21 ^b | 4/8 ^b |

^aStatistically significant trend for increased incidence by Cochran-Armitage test.

^bStatistically significantly different from controls according to Fisher's exact test performed for this review.

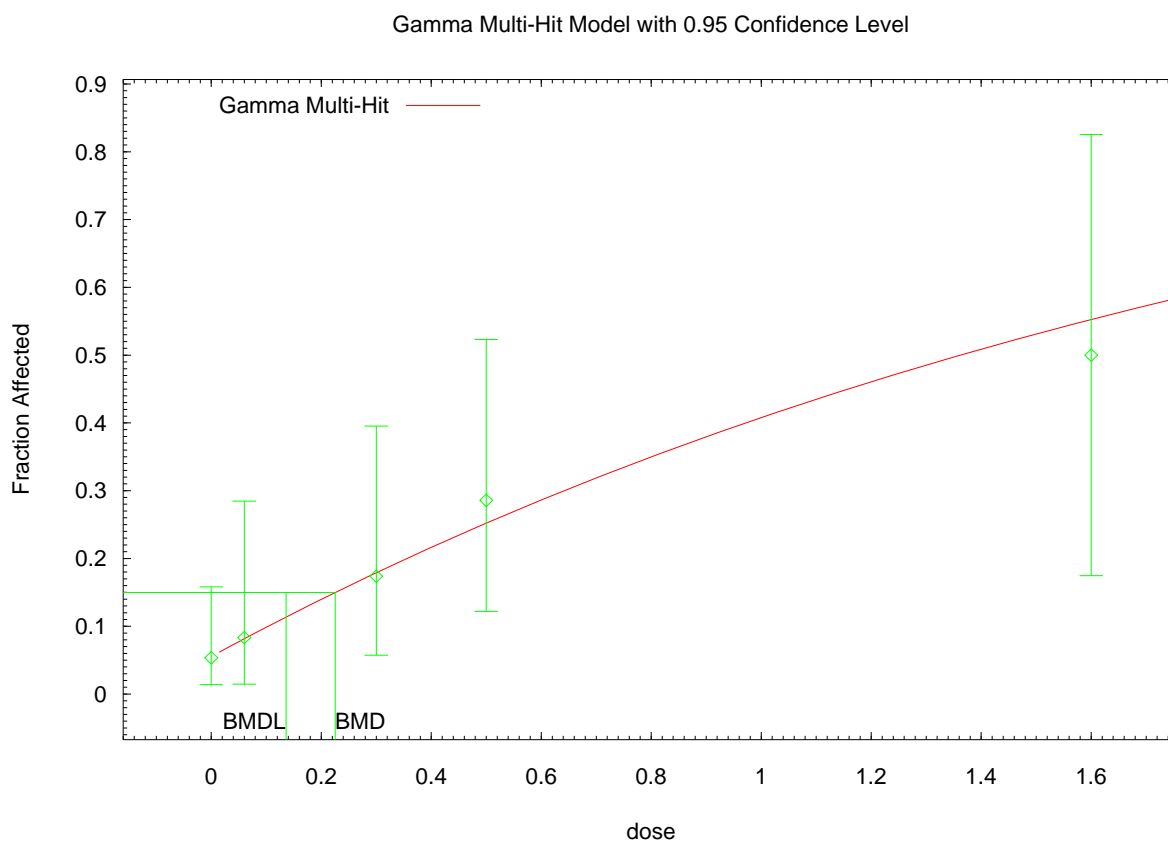
All models for dichotomous variables available in the U.S. EPA BMDS version 1.3.2 were fit to the data in Table B-5. All of the dichotomous models provided adequate fit to the data based on the summary results reported in the BMDS output and a more detailed examination of the graphs and chi-square goodness-of-fit statistics (summarized in Table B-6).

The gamma, multistage, and Weibull models yielded identical fits, and the lowest AIC (i.e., 98.9) value. Thus, these models were selected to calculate a potential POD for the RfD, based on the incidence data for liver lesions (fatty changes and hyperplasia) among rats. The model-predicted a BMD₁₀ associated with a 10% extra risk for liver lesions (fatty changes and hyperplasia) of 0.23 mg/kg-day. The BMDL₁₀ of 0.14 mg/kg-day was considered a potential POD for the RfD.

Table B-6. BMD modeling results for the increased incidence of liver lesions in rats (both sexes combined), following administration of chlordecone in the diet for 1–2 years

| Model | BMD ₁₀ | BMDL ₁₀ | χ^2 p-value | AIC |
|---|-------------------|--------------------|------------------|-------|
| Gamma, Multistage (1°) ^a , Weibull | 0.225 | 0.136 | 0.97 | 98.9 |
| Log-logistic | 0.200 | 0.106 | 0.95 | 100.7 |
| Probit | 0.327 | 0.217 | 0.74 | 99.9 |

^aMultistage model was run as 3rd degree polynomial with betas > 0.



16:22 04/20 2004

Figure B-3. Observed and predicted incidence of liver lesions in male and female Wistar rats following administration of chlordecone in the diet for 1–2 years.

Gamma Model of U.S. EPA BMDS (Version 1.3.2).

The computer output from the Gamma model of the incidence of liver lesions follows:

```
=====
$Revision: 2.2 $ $Date: 2001/03/14 01:17:00 $
Input Data File: C:\BMDS\LARSON_BOTHSEXES_DATA.(d)
Gnuplot Plotting File: C:\BMDS\LARSON_BOTHSEXES_DATA.plt
                        Tue Apr 20 16:22:31 2004
=====
```

BMDS MODEL RUN

The form of the probability function is:

$P[\text{response}] = \text{background} + (1 - \text{background}) * \text{CumGamma}[\text{slope} * \text{dose}, \text{power}]$,
where CumGamma(.) is the cumulative Gamma distribution function

Dependent variable = Frequency
Independent variable = Dose
Power parameter is restricted as power ≥ 1

Total number of observations = 5
Total number of records with missing values = 0
Maximum number of iterations = 250
Relative Function Convergence has been set to: 1e-008
Parameter Convergence has been set to: 1e-008

Default Initial (and Specified) Parameter Values
Background = 0.0614035
Slope = 0.901339
Power = 1.3

Asymptotic Correlation Matrix of Parameter Estimates

(*** The model parameter(s) -Power
have been estimated at a boundary point, or have been specified by the user,
and do not appear in the correlation matrix)

| | Background | Slope |
|------------|------------|-------|
| Background | 1 | -0.38 |
| Slope | -0.38 | 1 |

Parameter Estimates

| Variable | Estimate | Std. Err. |
|------------|-----------|-----------|
| Background | 0.0554334 | 0.0274998 |
| Slope | 0.467464 | 0.165121 |
| Power | 1 | NA |

NA - Indicates that this parameter has hit a bound implied by some inequality constraint and thus has no standard error.

Analysis of Deviance Table

| Model | Log(likelihood) | Deviance | Test DF | P-value |
|---------------|-----------------|----------|---------|----------|
| Full model | -47.3181 | | | |
| Fitted model | -47.428 | 0.219805 | 3 | 0.9743 |
| Reduced model | -54.3907 | 14.1451 | 4 | 0.006846 |

AIC: 98.8561

Goodness of Fit

| Dose | Est._Prob. | Expected | Scaled | | Residual |
|--------|------------|----------|----------|------|----------|
| | | | Observed | Size | |
| 0.0000 | 0.0554 | 3.104 | 3 | 56 | -0.06089 |
| 0.0600 | 0.0816 | 1.957 | 2 | 24 | 0.03177 |
| 0.3000 | 0.1790 | 4.118 | 4 | 23 | -0.06401 |
| 0.5000 | 0.2523 | 5.298 | 6 | 21 | 0.3525 |
| 1.6000 | 0.5529 | 4.423 | 4 | 8 | -0.3009 |

Chi-square = 0.22 DF = 3 P-value = 0.9737

Benchmark Dose Computation

Specified effect = 0.1

Risk Type = Extra risk

Confidence level = 0.95

BMD = 0.225388

BMDL = 0.136075

Decreased Epididymal Sperm Count in Male Rats Receiving Chlordecone in the Diet for 3 Months

Significantly decreased epididymal sperm count was observed in a 3-month feeding study in male Sprague-Dawley rats (Linder et al., 1983). Sperm count was significantly decreased according to ANOVA ($p < 0.05$) at the two highest doses tested (Table B-7).

Table B-7. Cauda Epididymal sperm count in male Sprague-Dawley rats receiving chlordecone in the diet for 3 months

| Dietary level (ppm) | 0 | 0.26 | 0.83 | 1.67 |
|--------------------------|--------------|--------------|---------------------------|---------------------------|
| Average dose (mg/kg-day) | 0 | 0.3 | 0.5 | 1.6 |
| Sperm count \pm SD | 308 \pm 44 | 290 \pm 32 | 248 \pm 70 ^a | 249 \pm 44 ^a |

^aStatistically different from controls according to ANOVA ($p < 0.05$).

Source: Linder et al. (1983).

All models for continuous variables available in the U.S. EPA BMDS version 1.4.1c, except the Hill model, were fit to the data in the table above. The Hill model was not fit to these data because fitting of the Hill model requires the estimation of four parameters (i.e., intercept, v , n , and k) which necessitates having a minimum of five dose groups in order to have adequate degrees of freedom for testing model fit. The Linder et al. (1983) study has only four dose groups, and thus the Hill model could not be fit to these data. All models fit were constant variance models. The default BMR, recommended for continuous data, of one estimated standard deviation from the control mean was selected (U.S. EPA, 2000c). The polynomial (2^o) model failed upon visual inspection. Specifically, the upturn in the curve near the high dose is not consistent with a monotonic change in the endpoint.

The linear and power models provided adequate fit to the decrease in epididymal sperm count based on the summary results reported in the BMDS output and a more detailed examination of the graphs and the chi-square goodness-of-fit statistics (summarized in Table B-8 and Figure B-4). These models yielded identical fits, essentially reducing the number of adequately fitting models to one. The BMD_{1SD} associated with a one standard deviation in the mean for decreased epididymal sperm count as 1.36 mg/kg-day (Table B-8). The lower 95% confidence limit on the benchmark dose ($BMDL_{1SD}$), a potential POD for the RfD, was 0.86 mg/kg-day (Table B-8).

Table B-8. BMD modeling results for decreased epididymal sperm count in rats, following administration of chlordecone in the diet for 3 months

| Model | BMD_{1SD} | $BMDL_{1SD}$ | χ^2 p -value | AIC |
|-------|-------------|--------------|---------------------|-----|
|-------|-------------|--------------|---------------------|-----|

| | | | | |
|-------------------------------|------|------|------|-------|
| Linear (1° polynomial), Power | 1.36 | 0.86 | 0.25 | 356.5 |
|-------------------------------|------|------|------|-------|

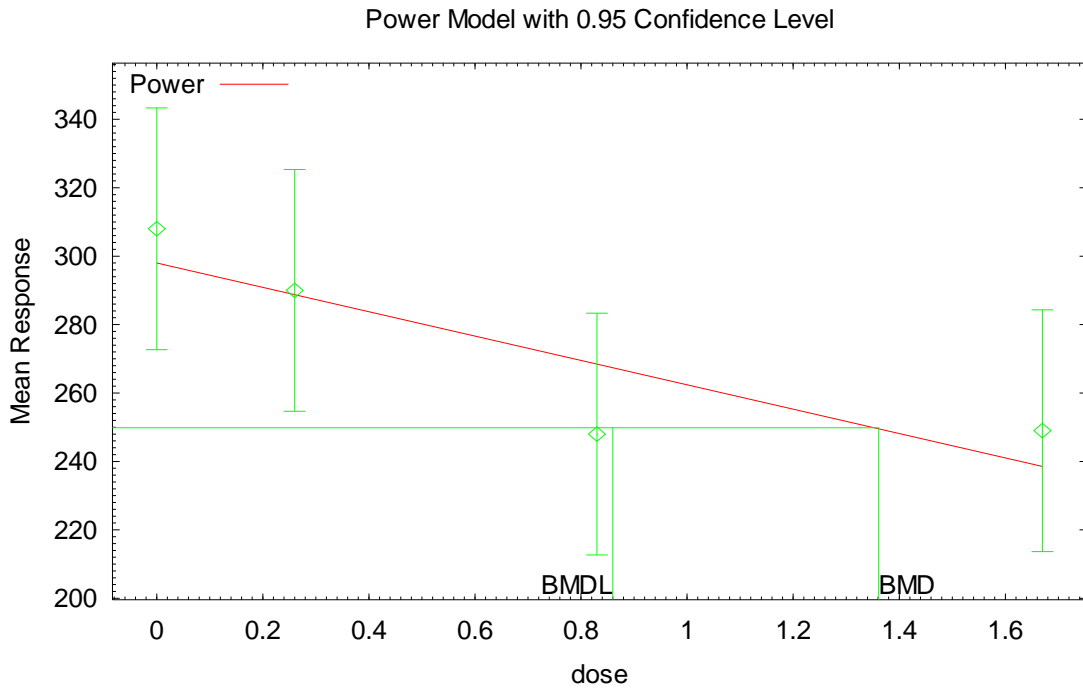


Figure B-4. Observed and predicted epididymal sperm count in male rats, following administration of chlordecone in the diet for 3 months.

Power Model of U.S. EPA BMDS (Version 1.4.1c).

The computer output from the Power model for this dataset follows:

```

=====
Power Model. (Version: 2.14; Date: 02/20/2007)
Input Data File: M:\_BMDS\SPERM-CONTENT.(d)
Gnuplot Plotting File: M:\_BMDS\SPERM-CONTENT.plt
Wed Jan 28 10:06:09 2009
=====

BMDS MODEL RUN
~~~~~

```

The form of the response function is:

$$Y[\text{dose}] = \text{control} + \text{slope} * \text{dose}^{\text{power}}$$

Dependent variable = MEAN

Independent variable = mg_kg_d

rho is set to 0

The power is restricted to be greater than or equal to 1

A constant variance model is fit

Total number of dose groups = 4
 Total number of records with missing values = 0
 Maximum number of iterations = 250
 Relative Function Convergence has been set to: 1e-008
 Parameter Convergence has been set to: 1e-008

Default Initial Parameter Values

alpha = 2439.87
 rho = 0 Specified
 control = 248
 slope = 2.80268
 power = -2.00961

Asymptotic Correlation Matrix of Parameter Estimates

(*** The model parameter(s) -rho -power
 have been estimated at a boundary point, or have been
 specified by the user, and do not appear in the correlation matrix)

| | alpha | control | slope |
|---------|----------|---------|----------|
| alpha | 1 | 1e-011 | 3.6e-011 |
| control | 1e-011 | 1 | -0.73 |
| slope | 3.6e-011 | -0.73 | 1 |

Parameter Estimates

| Variable | Estimate | Std. Err. | 95.0% Wald Confidence Interval | |
|----------|----------|-----------|--------------------------------|-------------------|
| | | | Lower Conf. Limit | Upper Conf. Limit |
| alpha | 2353.06 | 526.16 | 1321.8 | 3384.31 |
| control | 298.339 | 11.2737 | 276.243 | 320.435 |
| slope | -35.6368 | 11.9746 | -59.1067 | -12.167 |
| power | 1 | NA | | |

NA - Indicates that this parameter has hit a bound implied by some inequality constraint and thus has no standard error.

Table of Data and Estimated Values of Interest

| Dose | N | Obs Mean | Est Mean | Obs Std Dev | Est Std Dev | Scaled Res. |
|------|----|----------|----------|-------------|-------------|-------------|
| 0 | 10 | 308 | 298 | 44.3 | 48.5 | 0.63 |
| 0.26 | 10 | 290 | 289 | 31.6 | 48.5 | 0.0604 |
| 0.83 | 10 | 248 | 269 | 69.6 | 48.5 | -1.35 |
| 1.67 | 10 | 249 | 239 | 44.3 | 48.5 | 0.663 |

Model Descriptions for likelihoods calculated

Model A1: $Y_{ij} = \mu(i) + e(ij)$
 $\text{Var}\{e(ij)\} = \sigma^2$

Model A2: $Y_{ij} = \mu(i) + e(ij)$
 $\text{Var}\{e(ij)\} = \sigma(i)^2$

Model A3: $Y_{ij} = \mu(i) + e(ij)$
 $\text{Var}\{e(ij)\} = \sigma^2$
 Model A3 uses any fixed variance parameters that were specified by the user

Model R: $Y_i = \mu + e(i)$
 $\text{Var}\{e(i)\} = \sigma^2$

Likelihoods of Interest

| Model | Log(likelihood) | # Param's | AIC |
|--------|-----------------|-----------|------------|
| A1 | -173.886780 | 5 | 357.773561 |
| A2 | -170.660167 | 8 | 357.320335 |
| A3 | -173.886780 | 5 | 357.773561 |
| fitted | -175.269432 | 3 | 356.538863 |
| R | -179.269684 | 2 | 362.539369 |

Explanation of Tests

Test 1: Do responses and/or variances differ among Dose levels?
 (A2 vs. R)

Test 2: Are Variances Homogeneous? (A1 vs A2)

Test 3: Are variances adequately modeled? (A2 vs. A3)

Test 4: Does the Model for the Mean Fit? (A3 vs. fitted)

(Note: When $\rho=0$ the results of Test 3 and Test 2 will be the same.)

Tests of Interest

| Test | $-2 \cdot \log(\text{Likelihood Ratio})$ | Test df | p-value |
|--------|--|---------|----------|
| Test 1 | 17.219 | 6 | 0.008511 |
| Test 2 | 6.45323 | 3 | 0.09153 |
| Test 3 | 6.45323 | 3 | 0.09153 |
| Test 4 | 2.7653 | 2 | 0.2509 |

The p-value for Test 1 is less than .05. There appears to be a difference between response and/or variances among the dose levels. It seems appropriate to model the data.

The p-value for Test 2 is less than .1. Consider running a non-homogeneous variance model.

The p-value for Test 3 is less than .1. You may want to consider a

different variance model

The p-value for Test 4 is greater than .1. The model chosen seems to adequately describe the data

Benchmark Dose Computation

Specified effect = 1

Risk Type = Estimated standard deviations from the control mean

Confidence level = 0.95

BMD = 1.36119

BMDL = 0.859849

**APPENDIX C. TIME-TO-TUMOR MODELING RESULTS FROM TOX_RISK BASED
ON THE INCIDENCE OF HEPATOCELLULAR CARCINOMAS**

Male Osborne-Mendel Rats

Thu Oct 30 10:15:37 2008

Time-to_Tumor Input File: C:\Program Files\TOX_RISK\Kepone Male Rats Liver Tumors.ttd

Title: Chlordecone: Male Rats - Liver Tumors

Route/Dose Units: FOOD (ppm) Species : RAT
 Source : NCI 1976 Molecular WT. : 490.6
 Chemical : Chlordecone Weeks Of Study : 118
 # of Dose Group : 3 Dose Average Factor: 1
 ***** Dose Group 1 *****

of Dosing Periods : 1 Average Dose : 0.0
 ----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 118 Breathing Rate : .1805 l/min

===== Incidence Group 1 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|--------------|--------------|---------------|--------------|--------------|---------------|
| 96 | 1 | C | 115 | 5 | C |
| 118 | 4 | C | | | |

***** Dose Group 2 *****

of Dosing Periods : 4 Average Dose : 5.2
 ----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 6 Breathing Rate : .1805 l/min

----- Period 2 -----

Dose Level : 15 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 21 Breathing Rate : .1805 l/min

----- Period 3 -----

Dose Level : 5 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 59 Breathing Rate : .1805 l/min

----- Period 4 -----

Dose Level : 0 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 32 Breathing Rate : .1805 l/min

===== Incidence Group 2 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|--------------|--------------|---------------|--------------|--------------|---------------|
| 17 | 1 | C | 49 | 1 | C |
| 65 | 1 | C | 76 | 1 | C |
| 83 | 1 | C | 86 | 1 | C |
| 90 | 1 | C | 91 | 1 | C |
| 93 | 2 | C | 94 | 2 | C |
| 99 | 1 | C | 103 | 1 | C |
| 105 | 3 | C | 107 | 1 | C |
| 109 | 1 | C | 110 | 1 | C |
| 117 | 1 | C | 118 | 28 | C |
| 118 | 1 | I | | | |

***** Dose Group 3 *****

of Dosing Periods : 15 Average Dose : 15.7

----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 6 Breathing Rate : .1805 l/min

----- Period 2 -----

Dose Level : 30 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 55 Breathing Rate : .1805 l/min

----- Period 3 -----

Dose Level : 10 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 14 Breathing Rate : .1805 l/min

----- Period 4 -----

Dose Level : 10 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 1 Breathing Rate : .1805 l/min

----- Period 5 -----

Dose Level : 0 Adult Body Weight : 0.514 kg
 Days/Week : 7 Food Consump : 36 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 1 Breathing Rate : .1805 l/min

```

----- Period 6 -----
Dose Level : 10      Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 7 -----
Dose Level : 0      Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 8 -----
Dose Level : 10     Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 9 -----
Dose Level : 0      Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 10 -----
Dose Level : 10     Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 11 -----
Dose Level : 0      Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 12 -----
Dose Level : 10     Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 13 -----
Dose Level : 0      Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min
----- Period 14 -----
Dose Level : 10     Adult Body Weight : 0.514      kg
Days/Week  : 7      Food Consump      : 36      g/day
Hours/Day   : 24     Drinking Rate     : 35      ml/day
Duration    : 1      Breathing Rate    : .1805    l/min

```

----- Period 15 -----

| | | | | |
|------------|------|-------------------|---------|--------|
| Dose Level | : 0 | Adult Body Weight | : 0.514 | kg |
| Days/Week | : 7 | Food Consump | : 36 | g/day |
| Hours/Day | : 24 | Drinking Rate | : 35 | ml/day |
| Duration | : 32 | Breathing Rate | : .1805 | l/min |

===== Incidence Group 3 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 39 | 1 | U | 39 | 1 | U |
| 40 | 1 | C | 42 | 1 | C |
| 45 | 1 | C | 54 | 1 | U |
| 55 | 1 | C | 57 | 2 | C |
| 60 | 1 | C | 61 | 1 | C |
| 62 | 1 | C | 62 | 1 | U |
| 63 | 1 | C | 64 | 2 | C |
| 65 | 1 | C | 70 | 1 | U |
| 71 | 1 | C | 72 | 1 | U |
| 82 | 1 | C | 83 | 1 | C |
| 84 | 1 | C | 86 | 1 | C |
| 87 | 1 | C | 88 | 1 | C |
| 93 | 1 | C | 103 | 1 | C |
| 105 | 1 | C | 111 | 1 | C |
| 114 | 1 | I | 116 | 17 | C |
| 116 | 2 | I | | | |

***** NOTES *****

Generating Model Fit Table ---

TITLE: Chlordecone: Male Rats - Liver Tumors

Model: One Stage Weib Dataset: C:\Program
 Files\TOX_RISK\Kepone Male Rats Liver Tumors.ttd
 Functional form: 1 - EXP[(-Q0 - Q1 * D) * (T - T0)^Z]
 Maximum Log-Likelihood = -1.304794e+003

Parameter Estimates :

Q 0 = 0.000000E+000
 Q 1 = 0.000000E+000
 Z = 5.000000E+000
 T0 = 0.000000E+000 Set by User

| Avg. Doses (ppm) | ----- Number ----- | | |
|---------------------|--------------------|----------------------|---------------------------|
| | of animals | with fatal tumors | with incidental tumors |
| 0 | 10 | 0 | 0 |
| 5.1695 | 50 | 0 | 1 |
| 15.6780 | 50 | 0 | 3 |

Generating Extrapolated Doses Table ---

TITLE: Chlordecone: Male Rats - Liver Tumors

Dataset: C:\Program Files\TOX_RISK\Kepone Male Rats Liver
 Tumors.ttd

Exposure Pattern
 Model: One Stage Weib Age Begins: 0 Age Ends: 70
 Target Species: Human Weeks/Year: 52 Days/Week: 7
 Route: Food Hours/Day : 24
 Animal to human conversion method: MG/KG BODY WEIGHT(3/4)/DAY

Unit Potency [per mg/kg/day] (computed for Risk of 1.0E-6)
 Lower Bound = Not Reqstd MLE = Approaches 0 Upper Bound(q1*) =
 1.4944E+002

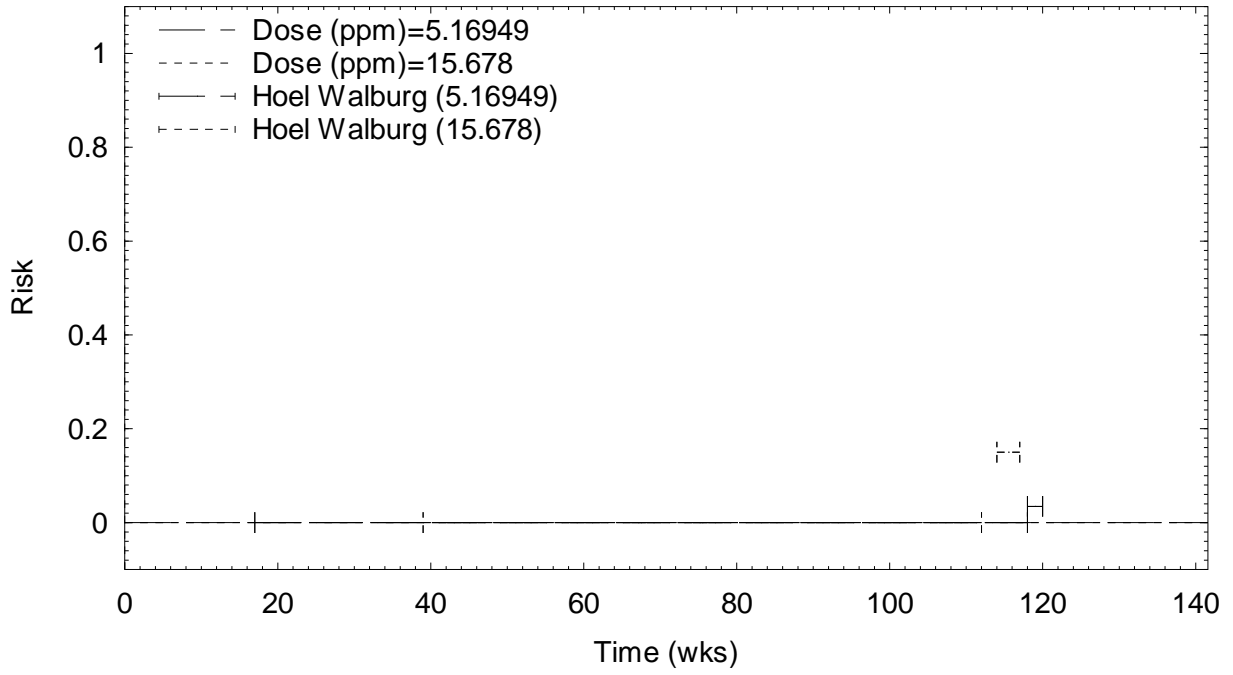
Induction Time (T0) Set by User to 0

| Incid Extra Risk | Time | Dose Estimates (ug/kg/day) | | |
|------------------|-------|----------------------------|-----------|------------------------|
| | | 95.00 % Lower Bound | MLE | 95.00 % Upper Bound |
| 1.0000E-006 | 70.00 | 6.6917E-006 | Unbounded | Not Reqstd |
| 1.0000E-005 | 70.00 | 6.7591E-005 | Unbounded | Not Reqstd |
| 0.0001 | 70.00 | 6.7594E-004 | Unbounded | Not Reqstd |
| 0.0010 | 70.00 | 6.7625E-003 | Unbounded | Not Reqstd |
| 0.01 | 70.00 | 6.7931E-002 | Unbounded | Not Reqstd |
| 0.10 | 70.00 | 7.1214E-001 | Unbounded | Not Reqstd |

09:54 11/05/2008

Incidental Graph

Kepona Male Rats Liver Tumors.ttd - Chlordecone: Male Rats - Liver Tumors
Model: One Stage Weib



Female Osborne-Mendel Rats

Thu Oct 30 11:37:44 2008

Time-to_Tumor Input File: C:\Program Files\TOX_RISK\Kepone Female Rats
Liver Tumors.ttd

Title: Chlordecone: Female Rats - Liver Tumors

Route/Dose Units: FOOD (ppm) Species : RAT
Source : NCI 1976 Molecular WT. : 490.6
Chemical : Chlordecone Weeks Of Study : 116
of Dose Group : 3 Dose Average Factor: 1

***** Dose Group 1 *****

of Dosing Periods : 1 Average Dose : 0.0
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.389 kg
Days/Week : 7 Food Consump : 30 g/day
Hours/Day : 24 Drinking Rate : 35 ml/day
Duration : 116 Breathing Rate : .1805 l/min

===== Incidence Group 1 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 66 | 1 | C | 92 | 1 | C |
| 110 | 1 | C | 115 | 7 | C |

***** Dose Group 2 *****

of Dosing Periods : 5 Average Dose : 12.2
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.389 kg
Days/Week : 7 Food Consump : 30 g/day
Hours/Day : 24 Drinking Rate : 35 ml/day
Duration : 6 Breathing Rate : .1805 l/min

----- Period 2 -----

Dose Level : 30 Adult Body Weight : 0.389 kg
Days/Week : 7 Food Consump : 30 g/day
Hours/Day : 24 Drinking Rate : 35 ml/day
Duration : 31 Breathing Rate : .1805 l/min

----- Period 3 -----

Dose Level : 15 Adult Body Weight : 0.389 kg
Days/Week : 7 Food Consump : 30 g/day
Hours/Day : 24 Drinking Rate : 35 ml/day
Duration : 24 Breathing Rate : .1805 l/min

----- Period 4 -----

Dose Level : 5 Adult Body Weight : 0.389 kg
Days/Week : 7 Food Consump : 30 g/day
Hours/Day : 24 Drinking Rate : 35 ml/day
Duration : 25 Breathing Rate : .1805 l/min

----- Period 5 -----

Dose Level : 0 Adult Body Weight : 0.389 kg
 Days/Week : 7 Food Consump : 30 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 30 Breathing Rate : .1805 l/min

===== Incidence Group 2 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 29 | 1 | C | 33 | 1 | C |
| 48 | 3 | C | 49 | 1 | C |
| 57 | 1 | U | 65 | 1 | C |
| 71 | 1 | C | 77 | 2 | C |
| 80 | 1 | C | 86 | 2 | C |
| 89 | 1 | I | 90 | 1 | C |
| 92 | 1 | C | 105 | 3 | C |
| 107 | 1 | C | 111 | 1 | C |
| 115 | 28 | C | | | |

***** Dose Group 3 *****

of Dosing Periods : 5 Average Dose : 17.9

----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.389 kg
 Days/Week : 7 Food Consump : 30 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 6 Breathing Rate : .1805 l/min

----- Period 2 -----

Dose Level : 60 Adult Body Weight : 0.389 kg
 Days/Week : 7 Food Consump : 30 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 6 Breathing Rate : .1805 l/min

----- Period 3 -----

Dose Level : 30 Adult Body Weight : 0.389 kg
 Days/Week : 7 Food Consump : 30 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 49 Breathing Rate : .1805 l/min

----- Period 4 -----

Dose Level : 10 Adult Body Weight : 0.389 kg
 Days/Week : 7 Food Consump : 30 g/day
 Hours/Day : 24 Drinking Rate : 35 ml/day
 Duration : 25 Breathing Rate : .1805 l/min

----- Period 5 -----

| | | | | |
|------------|------|-------------------|---------|--------|
| Dose Level | : 0 | Adult Body Weight | : 0.389 | kg |
| Days/Week | : 7 | Food Consump | : 30 | g/day |
| Hours/Day | : 24 | Drinking Rate | : 35 | ml/day |
| Duration | : 30 | Breathing Rate | : .1805 | l/min |

===== Incidence Group 3 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 11 | 1 | U | 26 | 1 | C |
| 30 | 1 | U | 34 | 1 | C |
| 45 | 1 | C | 48 | 1 | C |
| 57 | 1 | C | 59 | 1 | C |
| 65 | 1 | C | 71 | 1 | U |
| 75 | 2 | C | 78 | 1 | U |
| 81 | 1 | C | 84 | 1 | I |
| 85 | 1 | C | 86 | 1 | I |
| 90 | 1 | C | 101 | 4 | C |
| 101 | 1 | I | 102 | 2 | C |
| 105 | 2 | C | 105 | 2 | I |
| 109 | 1 | I | 115 | 2 | C |
| 116 | 14 | C | 116 | 4 | I |

***** NOTES *****

Generating Model Fit Table ---

TITLE: Chlordecone: Female Rats - Liver Tumors

Model: Two Stage Weib Dataset: C:\Program
 Files\TOX_RISK\Kepone Female Rats Liver Tumors.ttd
 Functional form: 1 - EXP[(-Q0 - Q1 * D - Q2 * D^2) * (T - T0)^Z]
 Maximum Log-Likelihood = -1.012917e+002

Parameter Estimates :

Q 0 = 0.000000E+000
 Q 1 = 0.000000E+000
 Q 2 = 1.240074E-024
 Z = 1.000000E+001
 T0 = 0.000000E+000 Set by User

| Avg. Doses (ppm) | Number | | |
|---------------------|------------|----------------------|---------------------------|
| | of animals | with fatal tumors | with incidental tumors |
| 0 | 10 | 0 | 0 |
| 12.1983 | 50 | 0 | 1 |
| 17.9310 | 50 | 0 | 10 |

Generating Extrapolated Doses Table ---

TITLE: Chlordecone: Female Rats - Liver Tumors

Dataset: C:\Program Files\TOX_RISK\Kepone Female Rats Liver
 Tumors.ttd

Exposure Pattern
 Model: Two Stage Weib Age Begins: 0 Age Ends: 70
 Target Species: Human Weeks/Year: 52 Days/Week: 7
 Route: Food Hours/Day : 24
 Animal to human conversion method: MG/KG BODY WEIGHT(3/4)/DAY

Unit Potency [per mg/kg/day] (computed for Risk of 1.0E-6)
 Lower Bound = Not Reqstd MLE = 1.1108E-003 Upper Bound(q1*) = 1.7507E+000

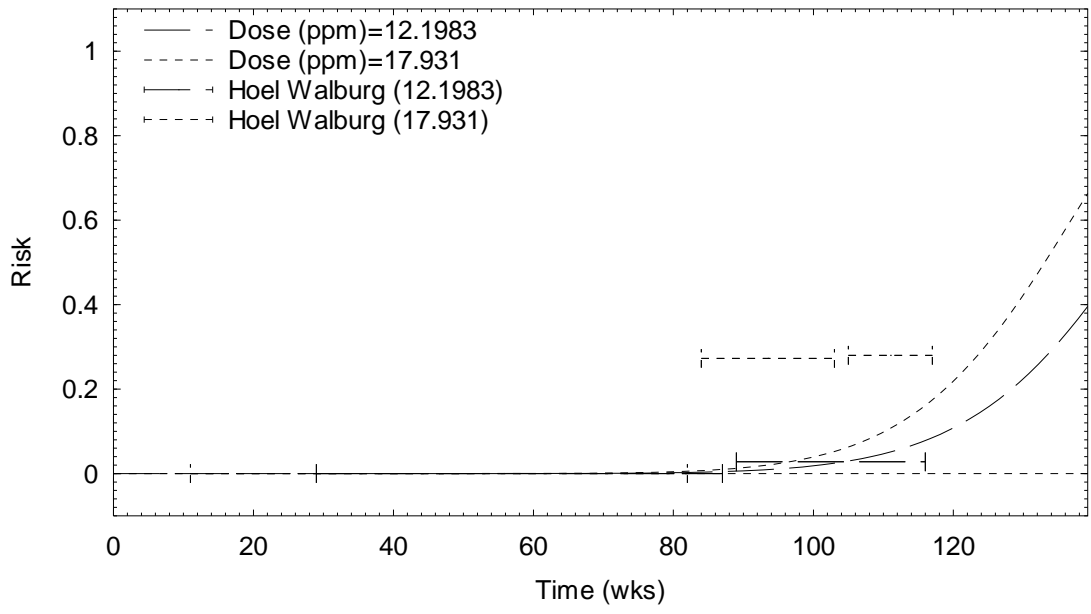
Induction Time (T0) Set by User to 0

| Incid Extra Risk | Time | Dose Estimates (ug/kg/day) | | |
|------------------|-------|----------------------------|-------------|------------------------|
| | | 95.00 % Lower Bound | MLE | 95.00 % Upper Bound |
| 1.0000E-006 | 70.00 | 5.7122E-004 | 9.0027E-001 | Not Reqstd |
| 1.0000E-005 | 70.00 | 5.7122E-003 | 2.8469E+000 | Not Reqstd |
| 0.0001 | 70.00 | 5.7122E-002 | 9.0029E+000 | Not Reqstd |
| 0.0010 | 70.00 | 5.7124E-001 | 2.8476E+001 | Not Reqstd |
| 0.01 | 70.00 | 5.7146E+000 | 9.0253E+001 | Not Reqstd |
| 0.10 | 70.00 | 5.7516E+001 | 2.9222E+002 | Not Reqstd |

11:52 11/06/2008

Incidental Graph

Kepono Female Rats Liver Tumors.ttd - Chlordecone: Female Rats - Liver Tumors
Model: Two Stage Weib



Male B6C3F₁ Mice

Tue Oct 28 15:20:02 2008

Time-to_Tumor Input File: C:\Program Files\TOX_RISK\Kepone Male Mice Liver Tumors.ttd

Title: Chlordecone: Male Mice - Liver Tumors

Route/Dose Units: FOOD (ppm) Species : MOUSE
Source : NCI 1976 Molecular WT. : 490.6
Chemical : Chlordecone Weeks Of Study : 95
of Dose Group : 3 Dose Average Factor: 1

***** Dose Group 1 *****

of Dosing Periods : 1 Average Dose : 0.0
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.0373 kg
Days/Week : 7 Food Consump : 6.4 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 95 Breathing Rate : .0347 l/min

===== Incidence Group 1 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|--------------|--------------|---------------|--------------|--------------|---------------|
| 66 | 1 | U | 92 | 1 | I |
| 95 | 13 | C | 95 | 5 | I |

***** Dose Group 2 *****

of Dosing Periods : 5 Average Dose : 16.8
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.0373 kg
Days/Week : 7 Food Consump : 6.4 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 6 Breathing Rate : .0347 l/min

----- Period 2 -----

Dose Level : 40 Adult Body Weight : 0.0373 kg
Days/Week : 7 Food Consump : 6.4 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 19 Breathing Rate : .0347 l/min

----- Period 3 -----

Dose Level : 20 Adult Body Weight : 0.0373 kg
Days/Week : 7 Food Consump : 6.4 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 23 Breathing Rate : .0347 l/min

----- Period 4 -----

Dose Level : 10 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 38 Breathing Rate : .0347 l/min

----- Period 5 -----

Dose Level : 0 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 9 Breathing Rate : .0347 l/min

===== Incidence Group 2 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 7 | 1 | C | 9 | 1 | U |
| 69 | 1 | U | 75 | 1 | I |
| 77 | 1 | C | 79 | 1 | C |
| 79 | 1 | I | 80 | 1 | I |
| 83 | 1 | I | 85 | 5 | I |
| 88 | 1 | I | 89 | 4 | I |
| 93 | 1 | C | 93 | 4 | I |
| 95 | 5 | C | 95 | 21 | I |

***** Dose Group 3 *****

of Dosing Periods : 4 Average Dose : 19.6

----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 6 Breathing Rate : .0347 l/min

----- Period 2 -----

Dose Level : 40 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 13 Breathing Rate : .0347 l/min

----- Period 3 -----

Dose Level : 20 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 67 Breathing Rate : .0347 l/min

----- Period 4 -----

Dose Level : 0 Adult Body Weight : 0.0373 kg
 Days/Week : 7 Food Consump : 6.4 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 9 Breathing Rate : .0347 l/min

===== Incidence Group 3 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 3 | 1 | C | 21 | 1 | C |
| 48 | 1 | C | 49 | 2 | C |
| 69 | 1 | I | 71 | 2 | I |
| 72 | 1 | U | 77 | 1 | I |
| 79 | 1 | C | 80 | 3 | I |
| 83 | 6 | I | 84 | 2 | I |
| 85 | 1 | I | 88 | 1 | I |
| 94 | 2 | I | 95 | 24 | I |

***** NOTES *****

Generating Model Fit Table ---

TITLE: Chlordecone: Male Mice - Liver Tumors

Model: One Stage Weib Dataset: C:\Program
Files\TOX_RISK\Kepone Male Mice Liver Tumors.ttd
Functional form: 1 - EXP[(-Q0 - Q1 * D) * (T - T0) ^ Z]
Maximum Log-Likelihood = -8.455136e+001

Parameter Estimates :

Q 0 = 5.156958E-021
Q 1 = 3.054341E-021
Z = 1.000000E+001
T0 = 0.000000E+000 Set by User

| Avg. Doses (ppm) | Number | | |
|---------------------|------------|----------------------|---------------------------|
| | of animals | with fatal tumors | with incidental tumors |
| 0 | 20 | 0 | 6 |
| 16.8421 | 50 | 0 | 39 |
| 19.5789 | 50 | 0 | 43 |

Generating Extrapolated Doses Table ---

TITLE: Chlordecone: Male Mice - Liver Tumors

Dataset: C:\Program Files\TOX_RISK\Kepone Male Mice Liver
Tumors.ttd

Exposure Pattern
Model: One Stage Weib Age Begins: 0 Age Ends: 70
Target Species: Human Weeks/Year: 52 Days/Week: 7
Route: Food Hours/Day : 24
Animal to human conversion method: MG/KG BODY WEIGHT(3/4)/DAY

Unit Potency [per mg/kg/day] (computed for Risk of 1.0E-6)
Lower Bound = Not Reqstd MLE = 7.0150E+000 Upper Bound(q1*) = 1.0422E+001

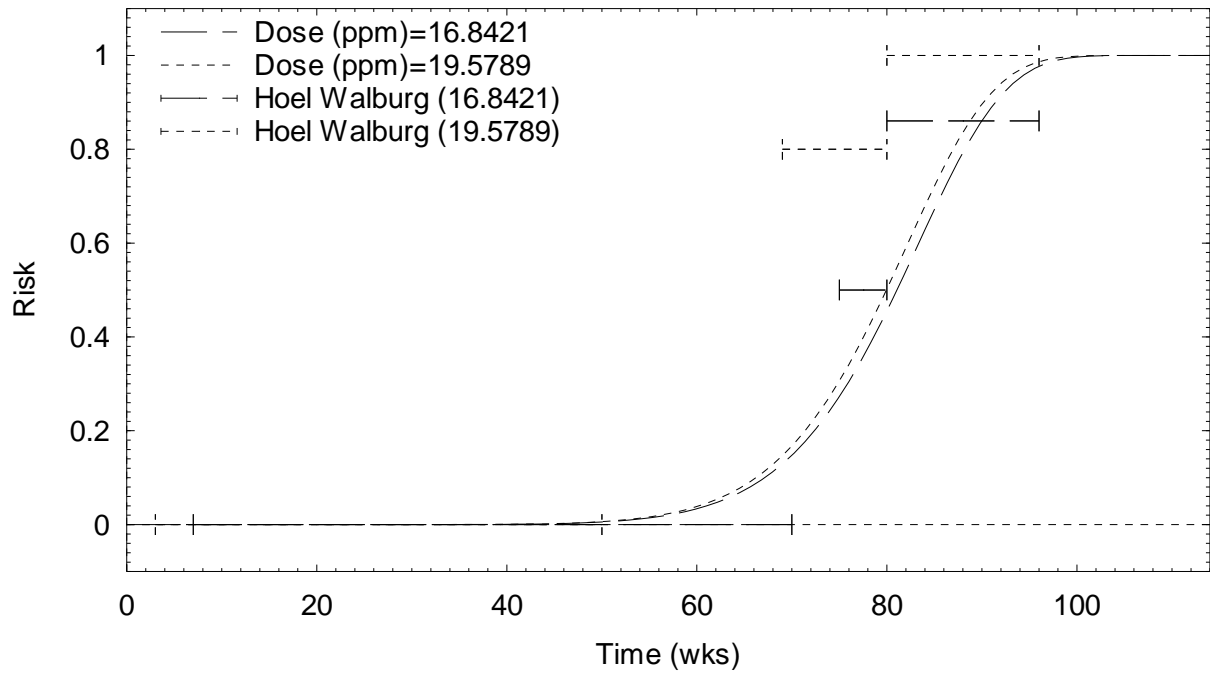
Induction Time (T0) Set by User to 0

| Incid Extra Risk | Time | Human Equivalent Dose Estimates (ug/kg/day) | | |
|------------------|-------|---|-------------|------------------------|
| | | 95.00 % Lower Bound | MLE | 95.00 % Upper Bound |
| 1.0000E-006 | 70.00 | 9.5955E-005 | 1.4255E-004 | Not Reqstd |
| 1.0000E-005 | 70.00 | 9.5955E-004 | 1.4255E-003 | Not Reqstd |
| 0.0001 | 70.00 | 9.5959E-003 | 1.4256E-002 | Not Reqstd |
| 0.0010 | 70.00 | 9.6003E-002 | 1.4262E-001 | Not Reqstd |
| 0.01 | 70.00 | 9.6438E-001 | 1.4327E+000 | Not Reqstd |
| 0.10 | 70.00 | 1.0110E+001 | 1.5019E+001 | Not Reqstd |

16:07 11/05/2008

Incidental Graph

Keponh Male Mice Liver Tumors.ttd - Chlordecone: Male Mice - Liver Tumors
Model: One Stage Weib



Female B6C3F₁ Mice

Tue Oct 28 15:21:04 2008

Time-to_Tumor Input File: C:\Program Files\TOX_RISK\Kepone Female Mice Liver Tumors.ttd

Title: Chlordecone: Female Mice - Liver Tumors

Route/Dose Units: FOOD (ppm) Species : MOUSE
Source : NCI 1976 Molecular WT. : 490.6
Chemical : Chlordecone Weeks Of Study : 95
of Dose Group : 3 Dose Average Factor: 1

***** Dose Group 1 *****

of Dosing Periods : 1 Average Dose : 0.0
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.0353 kg
Days/Week : 7 Food Consump : 6.1 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 95 Breathing Rate : .0347 l/min

===== Incidence Group 1 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|--------------|--------------|---------------|--------------|--------------|---------------|
| 93 | 1 | C | 95 | 9 | C |

***** Dose Group 2 *****

of Dosing Periods : 5 Average Dose : 16.8
----- Period 1 -----

Dose Level : 0 Adult Body Weight : 0.0353 kg
Days/Week : 7 Food Consump : 6.1 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 6 Breathing Rate : .0347 l/min

----- Period 2 -----

Dose Level : 40 Adult Body Weight : 0.0353 kg
Days/Week : 7 Food Consump : 6.1 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 19 Breathing Rate : .0347 l/min

----- Period 3 -----

Dose Level : 20 Adult Body Weight : 0.0353 kg
Days/Week : 7 Food Consump : 6.1 g/day
Hours/Day : 24 Drinking Rate : 6 ml/day
Duration : 23 Breathing Rate : .0347 l/min

----- Period 4 -----
 Dose Level : 10 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 38 Breathing Rate : .0347 l/min

----- Period 5 -----
 Dose Level : 0 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 9 Breathing Rate : .0347 l/min

===== Incidence Group 2 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|--------------|--------------|---------------|--------------|--------------|---------------|
| 71 | 1 | C | 89 | 1 | I |
| 93 | 2 | I | 95 | 23 | C |
| 95 | 23 | I | | | |

***** Dose Group 3 *****

of Dosing Periods : 5 Average Dose : 33.7

----- Period 1 -----
 Dose Level : 0 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 6 Breathing Rate : .0347 l/min

----- Period 2 -----
 Dose Level : 80 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 19 Breathing Rate : .0347 l/min

----- Period 3 -----
 Dose Level : 40 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 23 Breathing Rate : .0347 l/min

----- Period 4 -----
 Dose Level : 20 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 38 Breathing Rate : .0347 l/min

----- Period 5 -----
 Dose Level : 0 Adult Body Weight : 0.0353 kg
 Days/Week : 7 Food Consump : 6.1 g/day
 Hours/Day : 24 Drinking Rate : 6 ml/day
 Duration : 9 Breathing Rate : .0347 l/min

===== Incidence Group 3 =====

| Time (Weeks) | # Of Animals | Tumor Context | Time (Weeks) | # Of Animals | Tumor Context |
|-----------------|-----------------|------------------|-----------------|-----------------|------------------|
| 9 | 1 | C | 40 | 1 | C |
| 60 | 1 | U | 76 | 1 | I |
| 91 | 1 | I | 92 | 2 | C |
| 92 | 1 | I | 94 | 2 | I |
| 95 | 22 | C | 95 | 18 | I |

***** NOTES *****

Generating Model Fit Table ---

TITLE: Chlordecone: Female Mice - Liver Tumors

Model: One Stage Weib Dataset: C:\Program
Files\TOX_RISK\Kepone Female Mice Liver Tumors.ttd
Functional form: $1 - \text{EXP}[(-Q_0 - Q_1 * D) * (T - T_0)^Z]$
Maximum Log-Likelihood = -7.773364e+001

Parameter Estimates :

Q 0 = 0.000000E+000
Q 1 = 4.781306E-022
Z = 1.000000E+001
T0 = 0.000000E+000 Set by User

| Avg. Doses (ppm) | Number | | |
|---------------------|------------|----------------------|---------------------------|
| | of animals | with fatal tumors | with incidental tumors |
| 0 | 10 | 0 | 0 |
| 16.8421 | 50 | 0 | 26 |
| 33.6842 | 50 | 0 | 23 |

Generating Extrapolated Doses Table ---

TITLE: Chlordecone: Female Mice - Liver Tumors

Dataset: C:\Program Files\TOX_RISK\Kepone Female Mice Liver
Tumors.ttd

Exposure Pattern
Model: One Stage Weib Age Begins: 0 Age Ends: 70
Target Species: Human Weeks/Year: 52 Days/Week: 7
Route: Food Hours/Day : 24
Animal to human conversion method: MG/KG BODY WEIGHT(3/4)/DAY

Unit Potency [per mg/kg/day] (computed for Risk of 1.0E-6)
Lower Bound = Not Reqstd MLE = 1.1055E+000 Upper Bound(q1*) = 1.4561E+000

Induction Time (T0) Set by User to 0

Human Equivalent Dose Estimates (ug/kg/day)

| Incid Extra Risk | Time | 95.00 % | | Upper Bound |
|------------------|-------|-------------|-------------|-------------|
| | | Lower Bound | MLE | |
| 1.0000E-006 | 70.00 | 6.8678E-004 | 9.0457E-004 | Not Reqstd |
| 1.0000E-005 | 70.00 | 6.8679E-003 | 9.0457E-003 | Not Reqstd |
| 0.0001 | 70.00 | 6.8682E-002 | 9.0461E-002 | Not Reqstd |
| 0.0010 | 70.00 | 6.8713E-001 | 9.0502E-001 | Not Reqstd |
| 0.01 | 70.00 | 6.9024E+000 | 9.0912E+000 | Not Reqstd |
| 0.10 | 70.00 | 7.2360E+001 | 9.5306E+001 | Not Reqstd |

16:12 11/05/2008

Incidental Graph

Kepona Female Mice Liver Tumors.ttd - Chlordecone: Female Mice - Liver Tumors
Model: One Stage Weib

