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TOXICOLOGICAL REVIEW

OF

DICHLOROMETHANE (METHYLENE CHLORIDE)

(CAS No. 75-09-2)

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

December 2009

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(CAS No. 75-09-2)

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LIST OF ACRONYMS

ACGIH	American Conference of Governmental Industrial Hygienists
ADAF	age-dependent adjustment factor
AEGL	acute exposure guideline level
AIC	Akaike's Information Criterion
ALT	alanine aminotransferase
AP	alkaline phosphatase
AST	aspartate aminotransferase
ATSDR	Agency for Toxic Substances and Disease Registry
AUC	area under the curve of a concentration versus time plot
BAER	brainstem-auditory evoked response
BMD	benchmark dose
$BMDL_{10}$	95% lower bound on the BMD
BMR	benchmark response level
BW	body weight
CAEP	cortical-auditory-evoked potential
CASRN	Chemical Abstracts Service Registry Number
CHO	Chinese hamster ovary
CI	confidence interval
CMR	Chemical Marketing Reporter
CNS	central nervous system
COHb	carboxyhemoglobin
CV	coefficient of variation
CYP	cytochrome P450
DNA	deoxyribonucleic acid
EPA	U.S. Environmental Protection Agency
FEP	flash-evoked potential
FOB	functional observational battery
GD	gestation day
GSH	reduced glutathione
GST	glutathione S-transferase
HEC	human equivalent concentration
HPRT	hypoxanthine-guanine phosphoribosyl transferase
IARC	International Agency for Research on Cancer
ICD-9	International Classification of Diseases 9th ed.
IgM	immunoglobulin M
IRIS	Integrated Risk Information System
IUR	inhalation unit risk
LOAEL	lowest-observed-adverse-effect level
LOH	loss of heterozygosity
MCHC	mean corpuscular hemoglobin concentration
MCMC	Markov Chain Monte Carlo
Mg	milligrams
mRNA	messenger ribonucleic acid
NADPH	nicotinamide adenine dinucleotide phosphate
NIOSH	National Institute of Occupational Safety and Health
NLM	National Library of Medicine

NOAEL	no-observed-adverse-effect level
NRC	National Research Council
NTP	National Toxicology Program
OR	odds ratio
OSF	oral slope factor
OSHA	Occupational Safety and Health Administration
PBTK	physiologically based toxicokinetic
PND	postnatal day
QCC	cardiac output
RfC	reference concentration
RfD	reference dose
SD	standard deviation
SEM	standard error of the mean
SEP	somatosensory-evoked potential
SMR	standardized mortality ratio
SRC	Syracuse Research Corporation
SSB	single-strand break
TWA	time-weighted average
UF	uncertainty factor
VPR	ventilation:perfusion ratio

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 exposure to dichloromethane. It is not intended to be a comprehensive treatise on the chemical or toxicological nature of dichloromethane.

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 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 MANAGERS

Glinda S. Cooper, Ph.D.
Ambuja S. Bale, Ph.D., DABT
Office of Research and Development, IRIS Program
U.S. Environmental Protection Agency
Washington, DC

AUTHORS

Glinda S. Cooper, Ph.D.
Ambuja S. Bale, Ph.D., DABT
Andrew Rooney, Ph.D.
Paul Schlosser, Ph.D.
Allan Marcus, Ph.D.
Gene (Ching-Hung) Hsu, Ph.D., DABT
National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

John C. Lipscomb, Ph.D., DABT
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Cincinnati, OH

Peter McClure, Ph.D., DABT
Michael Lumpkin, Ph.D.
Fernando Llados, Ph.D.
Mark Osier, Ph.D., DABT
Daniel Plewak, B.S.
Syracuse Research Corporation
Syracuse, NY

Elizabeth Dupree Ellis, Ph.D.
Oak Ridge Institute for Science and Education
Center for Epidemiologic Research
Oak Ridge, TN

REVIEWERS

This document has been peer reviewed by EPA scientists.

INTERNAL EPA REVIEWERS

Ghazi Dannan, Ph.D
Karen Hogan, M.S.
Jennifer Jinot, Ph.D
Paul White, Ph.D
Samantha Jones, Ph.D.
Jamie Strong, Ph.D.
National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency

David Herr, Ph.D
National Health and Environmental Effect Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency

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 dichloromethane. 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. 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/m^3) 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 (extrarespiratory 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, an inhalation unit risk is a plausible upper bound on the estimate of risk per $\mu g/m^3$ air breathed.

Development of these hazard identification and dose-response assessments for dichloromethane has followed the general guidelines for risk assessment as set forth by the National Research Council (1983). 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, 1988a), *Guidelines for Developmental Toxicity Risk Assessment* (U.S. EPA, 1991), *Interim Policy for Particle Size*

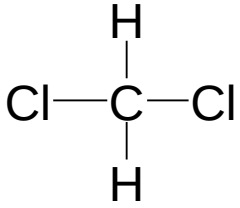
and Limit Concentration Issues in Inhalation Toxicity Studies (U.S. EPA, 1994a), *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994b), *Use of the Benchmark Dose Approach in Health Risk Assessment* (U.S. EPA, 1995), *Guidelines for Reproductive Toxicity Risk Assessment* (U.S. EPA, 1996), *Guidelines for Neurotoxicity Risk Assessment* (U.S. EPA, 1998a), *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 Risk 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 April 2009.

2. CHEMICAL AND PHYSICAL INFORMATION

Dichloromethane is a colorless liquid with a penetrating ether-like odor (Lewis, 1997).[1] Selected chemical and physical properties of dichloromethane are listed in Table 2-1.

Table 2-1. Physical properties and chemical identity of dichloromethane

	Physical property/chemical identity	Reference
CAS number	75-09-2	Lide (2000)
Synonyms	methylene chloride, methylene dichloride, methyl bichloride	O'Neil et al. (2001)
Molecular weight	84.93	O'Neil et al. (2001)
Chemical formula	CH_2Cl_2	O'Neil et al. (2001)
Boiling point	40°C	Lide (2000)
Melting point	−95.1°C	Lide (2000)
Vapor pressure	1.15×10^2 mm Hg at 25°C	Boublik et al. (1984)
Density	1.3266 g/mL at 20°C	Lide (2000)
Vapor density	2.93 (air = 1.02)	Holbrook (2003)
Water solubility	1.30×10^4 mg/L at 25°C	Horvath (1982)
Other solubility	Miscible in ethanol, ether, and dimethylformamide; soluble in carbon tetrachloride	International Agency for Research on Cancer (IARC) (1999)
Partition coefficient	log K_{ow} = 1.25	Hansch et al. (1995)
Flash point	Not flammable	U.S. Coast Guard (1999)
Auto ignition temperature	640°C	Holbrook (2003)
Latent heat of vaporization	3.30×10^5 J/kg	U.S. Coast Guard (1999)
Heat of fusion	16.89 cal/g	U.S. Coast Guard (1999)
Critical temperature	245.0°C	Holbrook (2003)
Critical pressure	6.171×10^6 Pa	Holbrook (2003)
Viscosity	0.430 cP at 20°C	Lewis (1997)
Henry's constant	3.25×10^{-3} atm m^3/mol at 25°C	Leighton and Calo (1981)
OH reaction rate constant	1.42×10^{-13} cm^3/molecule sec at 25°C	Atkinson (1989)
Chemical structure		

Dichloromethane is produced by two methods of manufacturing (International Agency for Research on Cancer [IARC], 1999). The older method involves the direct reaction of methane with chlorine either at high temperatures or at lower temperatures under catalytic or photolytic conditions (Holbrook, 2003). The more common method used today involves an initial reaction of hydrochloric acid with methanol to yield methyl chloride. Excess methyl chloride is then reacted in the gas phase thermally with chlorine to produce dichloromethane (Holbrook, 2003). This process can also be carried out catalytically or photolytically.

Dichloromethane became an important industrial chemical in the U.S. during World War II (Hardie, 1964). Dichloromethane has been used in paint strippers and removers, as a propellant in aerosols, in the manufacture of drugs, pharmaceuticals, film coatings, electronics, and polyurethane foam, and as a metal-cleaning solvent. Dichloromethane can also be used in the decaffeination process of coffee and tea (ATSDR, 2000). The U.S. production was 3.8 million pounds in 1941 and 8.3 million pounds in 1944 (Searles and McPhail, 1949). Dichloromethane production rose sharply in the decades following the war due to the increased demand for this substance for use mainly in paint strippers (Hardie, 1964; Searles and McPhail, 1949). U.S. production in 1947, 1955, 1960, and 1962 was approximately 19, 74, 113, and 144 million pounds, respectively (Hardie, 1964; Searles and McPhail, 1949). As other solvent uses and its use in aerosol propellants became important, demand for this substance increased further (Anthony, 1979). Dichloromethane production continued to rise dramatically through the 1970s; production capacities were 520 million pounds in 1973 and 830 million pounds in 1979 (Chemical Marketing Reporter [CMR], 1979, 1973).

After 1980, production of dichloromethane began to decline. Production capacities fell from 722 million pounds in 1982 to 465 million pounds in 1997 (CMR, 1997, 1982). The total U.S. production capacity for dichloromethane in 2000 was 535 million pounds (CMR, 2000). The demand for dichloromethane decreased from 600 million pounds in 1979 to 200 million pounds in 1999 (CMR, 2000, 1979). The decline in production of and demand for dichloromethane over the past 2 decades has been attributed to increased regulation, the use of alternative chemicals in aerosol spray cans, and concern over dichloromethane carcinogenicity (Holbrook, 2003; ATSDR, 2000).

Dichloromethane in the environment will partition mainly to air (National Library of Medicine [NLM], 2003). In air, dichloromethane exists as a vapor. Some of the dichloromethane released to soil or water is expected to volatilize to air. In soil, dichloromethane is expected to be highly mobile and may migrate to groundwater. The potential for dichloromethane to bioconcentrate in aquatic or marine organisms is low. Dichloromethane may biodegrade in soil or water under both aerobic and anaerobic conditions.

[1] To avoid confusion, "dichloromethane" is used throughout this summary even if a specific paper used the term "methylene chloride."

3. TOXICOKINETICS

3.1. ABSORPTION

3.1.1. Oral — Gastrointestinal Tract Absorption

There are currently no data available on absorption of dichloromethane following oral intake in humans. However, after oral administration in animals, dichloromethane is rapidly and nearly completely absorbed in the gastrointestinal tract (Angelo et al., 1986a, b; McKenna and Zempel, 1981). Angelo et al. (1986b) reported that, following administration of single radiolabeled oral doses (10, 50, or 200 mg/kg) to mature male F344 rats, 97% of the label was detected in the exhaled air within 24 hours, indicating nearly complete absorption. At several time points within 40 minutes of dose administration, less than 2% of the dose was found in the lower part of the gastrointestinal tract, indicating that the majority of dichloromethane absorption occurs in the upper gastrointestinal tract (Angelo et al., 1986b). Similar results were reported in mature male $B6C3F_1$ mice exposed to up to 50 mg/kg (Angelo et al., 1986a). In mature male Sprague-Dawley rats administered a single dose (1 or 50 mg/kg) of radiolabeled dichloromethane, less than 1% of the label was found in feces collected for 48 hours after dose administration (McKenna and Zempel, 1981). Absorption of dichloromethane generally follows first-order kinetics (Angelo et al., 1986a), and no evidence for a dichloromethane-specific carrier has been presented. The vehicle appears to affect the rate, but not the extent, of gastrointestinal absorption, with an aqueous vehicle resulting in a more rapid absorption of dichloromethane than an oil-based vehicle (Angelo et al., 1986a).

3.1.2. Inhalation—Respiratory Tract Absorption

Several studies in humans have demonstrated the absorption of dichloromethane following inhalation exposure. In a study by Astrand et al. (1975), 14 male volunteers (ages 19–29) were exposed to about 870 mg/m^3 (250 ppm) or 1,740 mg/m^3 (500 ppm) for 30 minutes while resting or exercising on a bicycle ergometer. There was a pause of about 20 minutes without exposure between rest and exercise periods. Uptake of dichloromethane was estimated at about 55% while resting and about 40, 30, and 35% at respective workloads of 50, 100, and 150 watts. Blood levels of dichloromethane correlated directly with exposure concentrations, and did not appear to increase when a workload was applied (Astrand et al., 1975). Similar reports of rapid uptake and a direct correlation between dichloromethane exposure level and blood levels in humans have been presented by other groups (DiVincenzo and Kaplan, 1981; DiVincenzo et al., 1971).

With extended (1–2 hours or greater) exposure, uptake tends to reach a steady-state level, at which point blood dichloromethane levels remain more or less constant (DiVincenzo and Kaplan, 1981; DiVincenzo et al., 1972; Riley et al., 1966). DiVincenzo et al. (1972) reported

that in humans exposed to 100 or 200 ppm of dichloromethane for 2 hours (without physical exercise), dichloromethane was rapidly absorbed, reaching an approximate steady state, as assessed by levels of unchanged dichloromethane in the expired air, within the first 15–30 minutes of exposure. A later study by the same group (DiVincenzo and Kaplan, 1981) similarly reported a rapid absorption of dichloromethane in volunteers exposed to 50–200 ppm for 7.5 hours on each of 5 consecutive days. A steady-state level, as assessed by levels of unchanged dichloromethane in the expired air, was reached quickly (1–2 hours), with exhaled dichloromethane levels increasing with increasing exposure level. A similar pattern was seen with blood dichloromethane levels. Estimated pulmonary uptake was 69–75% and did not vary appreciably with exposure concentration. In another experiment in which one of the investigators was seated during exposure to 100 ppm dichloromethane for 2 hours, concentrations of dichloromethane in expired air reached an apparent plateau of about 70 ppm within the first hour of exposure (Riley et al., 1966).

Body fat may influence absorption of dichloromethane, as evidenced by data from an experiment involving 12 men ages 21–35, divided into two groups (n = 6 per group) based on percent body fat (Engström and Bjurström, 1977). The mean percent body fat in the leaner group was 7.8% (standard error of the mean [SEM] 1.9), range 2.3–13.6%, compared with 25.1% (SEM 2.8), range 18.3–36.2%, in the more overweight group. Total uptake of dichloromethane during a light exercise period (50 watts[2]) for 1 hour with an exposure level of 750 ppm was positively correlated with percent body fat (r = 0.81), and the estimated amount of dichloromethane in fat storage was also correlated with percent body fat (r = 0.84).

A pattern of absorption similar to that seen in humans has been seen in animals. Initially, dichloromethane is readily absorbed following inhalation exposure, as evidenced by rapid appearance of dichloromethane in blood, tissues, and expired air (Withey and Karpinski, 1985; Stott and McKenna, 1984; Anders and Sunram, 1982; Carlsson and Hultengren, 1975; Roth et al., 1975). For example, absorption of inhaled 500 ppm dichloromethane in anesthetized, mature male F344 rats reached an apparent plateau within 10–20 minutes and was relatively constant for up to 2 hours (Stott and McKenna, 1984). In these experiments, absorption was calculated from measurements of exposure (nose only) and effluent concentrations and ventilation flow rate in intact animals; double tracheostomized rats were used to measure absorption in the isolated upper respiratory tract and the lower respiratory tract. At a ventilation rate of 53 mL/minute, absorption expressed as mean percentage of dichloromethane available for absorption was 44% (standard deviation [SD] 10) in intact rats, 13.2% (SD 3.6) in the upper respiratory tract, and 37% (SD 4.1) in the lower respiratory tract.

[2] A watt is the International System Unit of power and is equal to one joule of energy per second. It is a measure of the rate of energy use or production (i.e., the exercise effort that was exerted by the individuals in the study).

3.2. DISTRIBUTION

Results from studies of animals show that, following absorption, dichloromethane is rapidly distributed throughout the body and has been detected in all tissues that have been evaluated. Twenty minutes after a single intravenous dose of 10 mg [^{14}C]-dichloromethane/kg to mature male B6C3F$_1$ mice (Angelo et al., 1986a), total label was greatest in the liver (6.72 µg-equivalents/g tissue), with lower levels reported in the lung (1.82 µg-equivalents/g tissue), kidney (1.84 µg-equivalents/g tissue), and the remainder of the carcass (1.90 µg-equivalents/g tissue). By 4 hours post administration, levels in the liver had fallen to 3.08 µg-equivalents/g tissue, lung levels were 0.64 µg-equivalents/g tissue, and carcass levels were 0.23 µg-equivalents/g tissue. The levels in the kidney rose sharply in the first hour postexposure but then fell and remained steady at ~1.60 µg-equivalents/g tissue for the remaining 3 hours of the study (Angelo et al., 1986a). McKenna et al. (1982) exposed groups of mature male Sprague-Dawley rats to 50, 500, or 1,500 ppm [^{14}C]-labeled dichloromethane for 6 hours and examined tissues at 48 hours for presence of radiolabel; results are shown in Table 3-1. The greatest concentration of label was found in the liver, followed by the kidney and lung.

Table 3-1. Distribution of radioactivity in tissues 48 hours after inhalation exposure of mature male Sprague-Dawley rats (n = 3) for 6 hours

	Mean ± SD, µg-equivalent dichloromethane/g tissue, by exposure level		
Tissue	**50 ppm**	**500 ppm**	**1,500 ppm**
Liver	8.4 ± 1.5	35.6 ± 7.5	44.2 ± 3.5
Kidney	3.3 ± 0.1	16.2 ± 2.4	30.5 ± 0.2
Lung	1.9 ± 0.2	11.0 ± 1.3	16.5 ± 1.6
Brain	0.8 ± 0.3	4.2 ± 1.3	6.7 ± 0.2
Epididymal fat	0.5 ± 0.2	6.5 ± 0.5	4.1 ± 0.9
Skeletal muscle	1.1 ± 0.1	4.4 ± 1.9	7.7 ± 0.7
Testes	1.1 ± 0.2	5.5 ± 1.3	8.1 ± 0.5
Whole blood	1.1 ± 0.2	8.1 ± 1.9	8.9 ± 1.7
Remaining carcass	1.3 ± 0.2	5.9 ± 0.9	8.6 ± 1.4

Source: McKenna et al. (1982).

As noted in the preceding section, adipose tissue may affect the uptake of dichloromethane, and there is also evidence of a relation between adiposity and dichloromethane storage. In the study by Engström and Bjurström (1977) involving 12 men ages 21–35 exposed to 750 ppm dichloromethane during a 1 hour light exercise (50 watts) period, dichloromethane was measured in body fat biopsy specimens at 1, 2, 3, and 4 hours postexposure. All specimens were taken from the buttocks. The concentration of dichloromethane (per gram tissue) was

negatively correlated with percent body fat, but the total estimated amount of dichloromethane in fat tissue 4 hours postexposure was higher in subjects with a higher amount of fat (r = 0.84).

Carlsson and Hultengren (1975) exposed groups of 10 mature male Sprague-Dawley rats to [^{14}C]-dichloromethane for 1 hour at a mean concentration of 1,935 mg/m^3 (557 ppm) and SD of 90 mg/m^3 (26 ppm). The initial levels were highest in the white adipose tissue (approximately 80 µg dichloromethane per gram tissue) compared with approximately 35, 20, and 5 µg-equivalent dichloromethane/g tissue in the liver, kidney and adrenal glands, and brain, respectively. These initial levels in the adipose quickly fell to less than 10 µg-equivalent dichloromethane/g tissue; more moderate declines were seen in the other tissues.

With acute 6-hour exposure scenarios, peak exposure concentrations may have a greater influence on dichloromethane levels in the brain and perirenal fat than time-weighted average (TWA) concentrations during the exposure period (Savolainen et al., 1981). In rats exposed over a 6-hour period for 5 days/week to a TWA of 1,000 ppm dichloromethane consisting of two 1-hour peak concentrations (2,800 ppm) interspersed with exposure to 100 ppm, levels of dichloromethane in the brain and perirenal fat were significantly higher than corresponding levels in rats exposed to constant levels of 1,000 ppm. This difference was not seen with blood carbon monoxide (CO) levels (Table 3-2). With constant exposure concentrations of 500 or 1,000 ppm, perirenal fat levels of dichloromethane approximately doubled following 2 weeks of exposure compared with 1 week of exposure, indicating that some storage of dichloromethane in fat tissue can occur with repeated exposure scenarios (Table 3-2). In contrast, brain levels of dichloromethane in rats exposed for 1 week were higher than brain levels in rats exposed for 2 weeks. One possible explanation of these observations is that there is an induction of enzymes involved in dichloromethane metabolism in liver and other tissues with repeated exposure and dichloromethane in fat is poorly metabolized.

Table 3-2. Brain and perirenal fat dichloromethane and blood CO concentrations in male Wistar rats exposed by inhalation to dichloromethane at constant exposure concentrations compared with intermittently high exposure concentrations

	Exposure weeks					
Exposure level[a] (TWA, ppm)	**1**	**2**	**1**	**2**	**1**	**2**
	Brain (nmol/g)		**Perirenal fat (nmol/g)**		**Blood CO (nmol/g)**	
Control	0	0	0	0	40 ± 15	30 ± 10
500, constant	30 ± 7	9 ± 3	436 ± 47	918 ± 215	675 ± 195	781 ± 62
1,000, constant	33 ± 2	14 ± 3	1,316 ± 209	2,171 ± 219	876 ± 80	825 ± 56
1,000, with two 1-hour peaks of 2,800 ppm	111 ± 18	50 ± 15	2,295 ± 147	2,431 ± 146	728 ± 84	873 ± 90

[a]Groups of 5 rats were exposed to 0, 50, or 1,000 ppm 6 hours/day or 100 ppm interspersed with two 1-hour peaks of 2,800 ppm for 5 days/week for 1 or 2 weeks. Tissue concentration values are mean ± SD.

Source: Savolainen et al. (1981).

Placental transfer

Dichloromethane is capable of crossing the placental barrier and entering the fetal circulation. Anders and Sunram (1982) reported that when pregnant Sprague-Dawley rats (n = 3) were exposed to 500 ppm dichloromethane for 1 hour on gestational day (GD) 21, mean maternal blood levels were 176 nmol/mL (SEM 50), while fetal levels were 115 nmol/mL (SEM 40); interestingly, the levels of CO, a metabolite of dichloromethane, were similar in both the maternal blood (167 nmol/mL, SEM 12) and fetal blood (160 nmol/mL, SEM 31). Withey and Karpinski (1985) also reported higher maternal compared with fetal dichloromethane levels based on a study of five pregnant Sprague-Dawley rats exposed to 107–2,961 ppm of dichloromethane. Maternal blood levels of dichloromethane were 2–2.5-fold higher than those found in the fetal circulation.

Blood-brain barrier transfer

Dichloromethane is thought to readily transfer across the blood-brain barrier, as evidenced by the detection of radioactivity in brain tissue 48 hours after exposures of rats to radiolabeled dichloromethane at concentrations of 50, 500, or 1,500 ppm for 6 hours (McKenna et al., 1982) (see Table 3-1), and the historical demonstrations that dichloromethane has transient sedative and anesthetic properties in humans (for review of these reports, see Mattsson et al. [1990] and Winneke [1974]). Dichloromethane is no longer used as an anesthetic gas because the margin between anesthetic and lethal doses is narrow (Winneke, 1974).

3.3. METABOLISM

Metabolism of dichloromethane involves two primary pathways, outlined in Figure 3-1 (Agency for Toxic Substances and Disease Registry [ATSDR], 2000; Guengerich, 1997; Hashmi et al., 1994; Gargas et al., 1986). Dichloromethane is metabolized to CO in a cytochrome P450 (CYP)-dependent oxidative pathway that is predominant at low exposure levels. The CYP-related pathway results in the addition of oxygen, followed by spontaneous rearrangement to formyl chloride, and then to CO; each spontaneous rearrangement releases H^+ and Cl^- ions. At higher exposure levels, the CYP pathway becomes saturated and a second pathway begins to predominate. Glutathione S-transferase (GST)-catalyzed addition of glutathione (GSH) is the initial step in this pathway. The replacement of one of the chlorine atoms with the S-glutathione group results in formation of S-(chloromethyl)glutathione and the release of H^+ and Cl^- ions. Hydration of S-(chloromethyl)glutathione results in an S-glutathionyl methanol molecule, which can spontaneously form formaldehyde or rearrange to form an S-glutathione formaldehyde molecule, and then further rearrange to formate. Both formaldehyde and formate can then be further metabolized to CO_2.

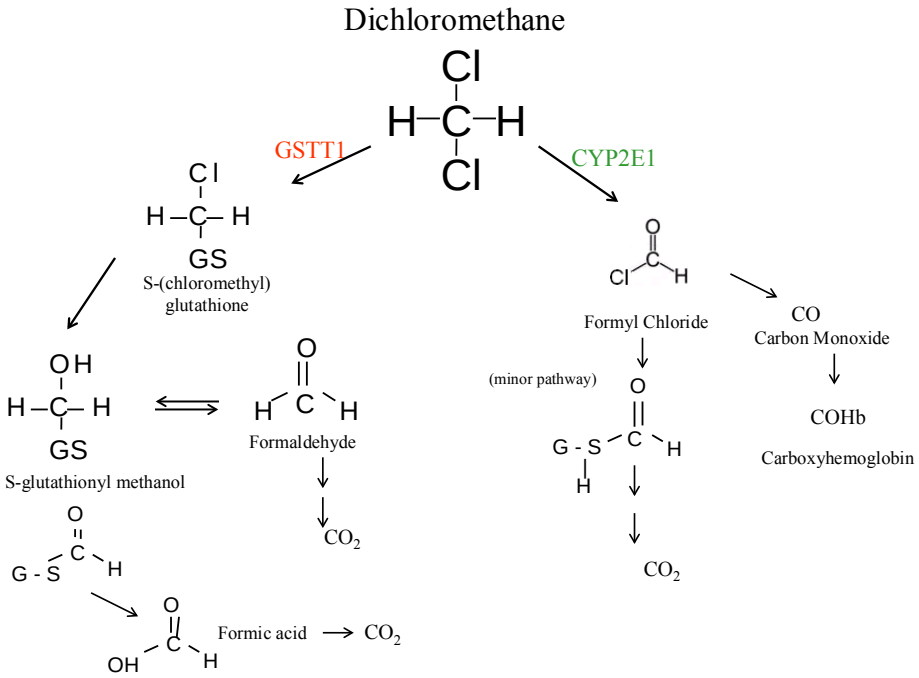


Figure 3-1. Proposed pathways for dichloromethane metabolism.

Adapted from: ATSDR (2000); Guengerich (1997); Hashmi et al. (1994); Gargas et al. (1986).

As described in the following discussion of the two pathways, a metabolic balance appears to exist between them, with the CYP pathway tending to be relatively more active at lower doses and the GST pathway metabolizing the majority of a dichloromethane dose at higher exposure levels once the CYP pathway has become saturated. Exposure to other agents may shift this balance. For example, pretreatment with compounds that deplete GSH (e.g., buthionine sulfoximine, diethylmaleate, phorone) resulted in an increase in blood carboxyhemoglobin (COHb) levels, following a single injection of dichloromethane, relative to animals that did not receive GSH depletion, indicating a shift to the CYP pathway (Oh et al., 2002). Similarly, co-exposure to agents that compete for CYP2E1 results in a shift toward the GST pathway and away from CO production (Lehnebach et al., 1995; Pankow and Jagielki, 1993; Pankow et al., 1991a, b; Glatzel et al., 1987; Roth et al., 1975).

3.3.1. The CYP2E1 Pathway

There is considerable evidence of the importance of the CYP2E1 metabolic pathway in studies in animals (Oh et al., 2002; Wirkner et al., 1997; Kim and Kim, 1996; Lehnebach et al., 1995; Pankow et al., 1991a, b; Pankow and Hoffmann, 1989; Pankow, 1988; Glatzel et al., 1987; Angelo et al., 1986a, b; Landry et al., 1983; Anders and Sunram, 1982; McKenna et al., 1982; McKenna and Zempel, 1981; Rodkey and Collison, 1977; Carlsson and Hultengren, 1975; Roth et al., 1975; Fodor et al., 1973) and humans (Takeshita et al., 2000; DiVincenzo and Kaplan, 1981; Astrand et al., 1975). These studies demonstrate that exposure to dichloromethane, regardless of exposure route, results in the formation of CO, as assessed by direct measurements of elevated levels of CO in expired air and increased levels of COHb in the blood.

The first step in the CYP2E1 pathway is the formation of formyl chloride (Figure 1). Watanabe and Guengerich (2006) conducted a series of studies to investigate the downstream metabolites of formyl chloride, and reported only marginal (3% maximum at pH 9) formation of *S*-formyl GSH from formyl chloride in the presence of GSH. Therefore, most (>97%) of the formyl chloride is metabolized further to carbon monoxide. Furthermore, CO formation from formyl chloride was independent of GSH presence in the assay.

Results from numerous studies in rats in which CYP2E1 metabolism was blocked or induced indicate that the generation of CO occurs as a result of metabolism of dichloromethane by the CYP2E1 pathway (Figure 3-1). Co-exposure of rats to a high dose of ethanol (174 mmol/kg), which is metabolized by CYP2E1, and dichloromethane (1.6, 6.2, 15.6 mmol/kg) resulted in no increase in blood COHb, indicating that the metabolic pathway for CO formation had been either blocked or saturated (Glatzel et al., 1987). Similar results have been seen with coadministration of other known CYP substrates, including diethyldithiocarbamate (Lehnebach et al., 1995), methanol (Pankow and Jagielki, 1993), benzene, toluene, and three xylene isomers (Pankow et al., 1991b). Pretreatment of animals with CYP inducers (e.g., benzene, toluene, xylenes, methanol, isoniazid), particularly those that induce CYP2E1, resulted in an increased level of CO formation, as assessed by COHb formation or measurement in expired air, following single exposures to dichloromethane (Kim and Kim, 1996; Pankow and Jagielki, 1993; Pankow et al., 1991b; Pankow and Hoffmann, 1989; Pankow, 1988). Pretreatment with disulfuram, a CYP2E1 blocker, resulted in a complete lack of formation of COHb following dichloromethane exposure, indicating that CYP2E1 is the isozyme responsible for metabolism of dichloromethane (Kim and Kim, 1996).

Evidence in hamster and rat studies suggests that the CYP2E1 pathway becomes saturated at high dichloromethane exposure levels; comparable data from studies in mice were not found. In hamsters, mean COHb percentages were elevated to a similar degree (about 28–30%, compared with <1% in controls) in three groups exposed by inhalation to 500, 1,500, or 3,500 ppm dichloromethane for 6 hours (Burek et al., 1984). After 21 months of exposure by this protocol, mean COHb percentages in the three exposure groups remained similarly elevated,

indicative of saturation of the CYP2E1 pathway in hamsters at exposure levels ≥500 ppm and a lack of accumulation of dichloromethane and CYP2E1 metabolites with chronic exposure. McKenna et al. (1982) found that blood COHb levels in rats increased when inhalation exposure concentration was increased from 50 to 500 ppm but that similar levels of COHb were reported following exposure to 1,500 ppm as following exposure to 500 ppm; the peak blood COHb percentages were approximately 10%. In rats exposed to 0, 50, 200, or 500 ppm for 6 hours/day, 5 days/week for 2 years, mean COHb percentages were 2.2, 6.5, 12.5, and 13.7%, respectively, suggesting that saturation of the CYP2E1 pathway is approached at 200 ppm (Nitschke et al., 1988a). In male F344 rats exposed for 4 hours to dichloromethane concentrations of about 150, 300, 600, 1,000, and 2,000 ppm, mean COHb percentages (estimated from a figure) were about 4% at 150 ppm and about 8% at each of the four higher exposure concentrations (Gargas et al., 1986). McKenna and Zempel (1981) reported that increasing the oral dose of labeled dichloromethane from 1 mg/kg to 50 mg/kg in rats resulted in a lower fraction of the total dose being metabolized to CO. Single injections of 3 and 6 mmol/kg of dichloromethane in rats resulted in nearly identical levels of blood COHb (Oh et al., 2002).

In human subjects exposed to dichloromethane in the workplace, saturation of CYP metabolism appears to be approached in the 400–500 ppm range (Ott et al., 1983e). Blood samples were drawn during working hours from 136 fiber production workers who were exposed to dichloromethane, acetone, and methanol. TWA exposure concentrations for the workers were determined by personal monitoring techniques, and percent COHb levels in the blood samples were determined. Estimated TWA concentrations in the exposed workers showed a bimodal distribution. The lower mode of exposure concentrations showed the highest frequency in the 150–200 ppm range, while the higher mode showed the highest frequency in the range of 450–500 ppm. Plots of percent COHb against TWA exposure concentrations showed that saturation begins to be apparent in the 400–500 ppm range of exposure concentrations.

The liver is the tissue most enriched in CYP2E1 catalytic activity, but CYP2E1 protein and messenger ribonucleic acid (mRNA) have been detected in other human tissues, including the lung, brain, kidney, pancreas, bladder, small intestine, and blood lymphocytes (Nishimura et al., 2003). As such, the liver is expected to be the main site of CYP metabolism of dichloromethane, but other tissues are also expected to metabolize dichloromethane via this pathway. Of particular relevance given the neurologic effects seen with dichloromethane are the distribution and inducibility of CYP2E1 in different areas of the brain (Miksys and Tyndale, 2004). Individuals with decreased CYP2E1 activity may experience decreased generation of CO and an increased level of GST-related metabolites following exposure to dichloromethane. As a result, these individuals may be more susceptible to the chronic effects of dichloromethane from GST-related metabolites than individuals with higher levels of CYP2E1 activity. Conversely, individuals with higher CYP2E1 activity may experience relatively increased generation of CO

at a given dichloromethane exposure level and, therefore, may be more susceptible to the acute toxicity of dichloromethane (from CO).

Results from studies examining human interindividual variation in CYP2E1 activities (e.g., catalytic activities, protein levels, or mRNA levels) indicate that individuals may vary in their ability to metabolize dichloromethane through the CYP2E1 pathway. In a study of liver samples from 30 Japanese and 30 Caucasian individuals, two- to threefold variation was found in the levels of CYP2E1 protein, whereas catalytic activity toward substrates associated with CYP2E1 (e.g., 7-ethoxycoumarin) displayed a wider range of values, approximately 25-fold; no clear gender-specific or ethnic differences were found in hepatic levels of CYP2E1 protein or enzymatic activities associated with CYP2E1 (Shimada et al., 1994). In a study of interindividual variation in 70 healthy human subjects (40 men and 30 women) given an oral dose of chlorzoxazone, a therapeutic agent whose metabolism and blood clearance has been related to CYP2E1 levels, a three- to fourfold range in plasma half-life and clearance values was observed, with no clear or dramatic age- or gender-specific differences (Kim et al., 1995). A six- to sevenfold range in chlorzoxazone hydroxylation activity was reported for a group of 69 healthy, smoking and nonsmoking male and female volunteers with mixed ethnic backgrounds; the range was markedly increased when a group of 72 alcoholic inpatients was included (Lucas et al., 1999). In studies of human liver microsomes, four- to sixfold ranges in CYP2E1-dependent oxidation of trichloroethylene have been reported (Lipscomb et al., 2003, 1997). CYP2E1 protein levels in 50 specimens of human lymphocytes from healthy individuals showed an approximate fivefold range (Bernauer et al., 2000), and a 3.7-fold range in liver CYP2E1 mRNA levels was reported for a group of 24 patients with chronic hepatitis (Haufroid et al., 2003). More recently, a threefold range was reported for maximal rates of hepatic CYP2E1-catalyzed metabolism of dichloromethane, which were estimated with a modified physiologically based toxicokinetic (PBTK) model originally developed by Andersen et al. (1987) and kinetic data (e.g., dichloromethane breath and blood concentrations) for 13 volunteers (10 males and 3 females) exposed to one or more concentrations of dichloromethane by inhalation for 7.5 hours (Sweeney et al., 2004). In summary, most studies indicate a three- to sevenfold variability in CYP2E1 activity, as assessed by various types of measurements, among "healthy" volunteers. However, various clinical factors (i.e., obesity, alcoholism, use of specific medications) or co-exposures (i.e., to various solvents) (Lucas et al., 1999) may result in greater variation, and thus the potential for saturation at lower exposures, within the general population.

Several genetic polymorphisms for the human CYP2E1 gene have been described, but clear and consistent correlations with interindividual variation in CYP2E1 protein levels or associated enzyme activities have not been identified (Ingelman-Sundberg, 2004; Lucas et al., 2001; Kim et al., 1995; Shimada et al., 1994). The most frequently studied CYP2E1 polymorphisms, *RsaI/PstI*, are located in the 5′-flanking region of the gene, and mutations are thought to lead to increased CYP2E1 protein expression via transcription (Lucas et al., 2001).

Available data indicate that the frequency of this polymorphism, as well as other CYP2E1 polymorphisms, varies among ethnic groups. For example, Stephens et al. (1994) examined blood samples from 126 African-Americans, 449 European Americans, and 120 Taiwanese subjects and found frequencies for a rare *RsaI* allele (C2) of 0.01 in African-Americans, 0.04 in European Americans, and 0.28 in Taiwanese subjects. In a study of 102 Mexicans, the reported mutation frequency at the *RsaI* C2 allele was 0.30 (Mendoza-Cantú et al., 2004).

3.3.2. The GST Pathway

The other major pathway for dichloromethane metabolism involves the conjugation of dichloromethane to GSH, catalyzed by GST. This results in the formation of a GSH conjugate that is eventually metabolized to CO_2 (Figure 3-1). The conjugation of dichloromethane to GSH results in formation of two reactive intermediates that have been proposed to be involved in dichloromethane toxicity, S-(chloromethyl)glutathione and formaldehyde. In studies with rat, mouse, and human liver cytosol preparations in the presence of GSH, examination of metabolites with ^{13}C-NMR indicated that S-(chloromethyl)glutathione was an intermediate in the pathway to formaldehyde (Hashmi et al., 1994). Formaldehyde formation from dichloromethane has been noted in human (Bruhn et al., 1998; Hallier et al., 1994; Hashmi et al., 1994), rat, and mouse (Casanova et al., 1997; Hashmi et al., 1994) cells in vitro. Formation of free hydrogen ion is also hypothesized, although no direct evidence supporting this has been presented.

The GST pathway has approximately a 10-fold lower affinity for dichloromethane than the CYP pathway (Reitz et al., 1989; Andersen et al., 1987). At lower exposure concentrations, the CYP pathway is expected to predominate, but, as exposure concentrations increase, the GST pathway is expected to gain in relative importance as a dispositional pathway for absorbed dichloromethane. Based on in vitro studies with liver preparations, the estimated Michaelis-Menten kinetic constants (K_ms) in GST assays with dichloromethane were about 137 mM in a B6C3F$_1$ mouse preparation and about 44 mM in two human preparations (Reitz et al., 1989). In contrast, estimated K_ms in CYP assays were about 1.8, 1.4, and 2.0 mM in B6C3F$_1$ mouse, F344 rat, and Syrian golden hamster preparations, respectively. In four human liver preparations, estimated CYP K_ms were about 2.6, 2.0, 0.9, and 2.8 mM (Reitz et al., 1989).

Early investigations indicated that in humans GSTs of the α-, μ-, and π-classes were not responsible for the metabolism of dichloromethane (Bogaards et al., 1993). Tissue samples that metabolized substrates specific to those GST classes did not conjugate dichloromethane to GSH. Later investigations identified the recently-characterized GST theta class (Meyer et al., 1991), specifically GST-theta1-1 (GST-T1), as the GST isoenzyme responsible for the metabolism of dichloromethane (Mainwaring et al., 1996; Blocki et al., 1994). In the absence of the GST-T1 gene, no deoxyribonucleic acid (DNA)-protein cross-links were formed by human liver cells exposed to dichloromethane (Casanova et al., 1997), and formaldehyde production was not detected in human erythrocytes (Hallier et al., 1994). In a mouse model with a disrupted GST-

T1 gene, GST activity with dichloromethane in liver and kidney cytosol samples was substantially lower compared with wild-type GST mice (Fujimoto et al., 2007).

A polymorphism of the GST-T1 gene has been demonstrated in humans. People with two functional copies of the gene (+/+) readily conjugate GSH to dichloromethane. Individuals having only one working copy of the gene (+/–) display relatively decreased conjugation ability. Individuals with no functional copy of the gene (–/–) do not express active GST-T1 protein and do not metabolize dichloromethane via a GST-related pathway (Thier et al., 1998). Results from studies of GST-T1 genotypes in human blood samples indicate that average prevalences of the GST-T1 null (–/–) genotype are higher in Asian ethnic groups (47–64%) than in other groups, including Caucasians (19–20%), African-Americans (22%), and mixed groups (19%) (Raimondi et al., 2006; Garte et al., 2001; Nelson et al., 1995) (see Table 3-3). Although information on the age distribution of study subjects was not generally reported in these analyses, there is little reason to expect effect modification by age since this is not a gene linked to early mortality. Based on data collected by Nelson et al. (1995) and U.S. 2000 census data (and assuming Hardy-Weinberg equilibrium), Haber et al. (2002) calculated U.S. average distributions of GST-T1 genotypes as follows: 32% +/+; 48% +/–; and 20% –/–.

Table 3-3. Mean prevalences of the GST-T1 null (–/–) genotype in human ethnic groups

	Reference		
Ethnic group	**Nelson et al. (1995)[a]**	**Garte et al. (2001)[b]**	**Raimondi et al. (2006)[c]**
Chinese	64.4% (n = 45)	Not reported	Not reported
Korean	60.2% (n = 103)	Not reported	Not reported
Caucasian	20.4% (n = 442)	19.7% (n = 5,577)	19.0% (n = 6,875)
Asian	Not reported	47.0% (n = 575)	53.6% (n = 1,727)
African-American	21.8% (n = 119)	Not reported	Not reported
Mexican American	9.7% (n = 73)	Not reported	Not reported
Other	Not reported	Not reported	19.4 % (n = 1,485)

[a]Nelson et al. (1995) examined prevalence of the null GST-T1 genotype from analysis of blood samples from subjects of various ethnicities as noted above.
[b]Garte et al. (2001) collected GST-T1 genotype data in Caucasian (29 studies; 5,577 subjects) and Asian (3 studies, 575 subjects) ethnic groups; subjects were controls in case-control studies of cancer and various polymorphisms in genes for bioactivating enzymes.
[c]Raimondi et al. (2001) collected GST-T1 genotype data from 35 case-control studies of cancer and GST-T1 genotype; data in this table are for control subjects. The "other" group in this study is defined as Latino, African-American, and mixed ethnicities.

Results from a study of the distribution of activity levels for in vitro conjugation of dichloromethane with GSH in 22 human liver samples are roughly reflective of these estimates of the distribution of this polymorphism (Bogaards et al., 1993). No activity was found in 3/22 of the liver samples. Eleven of the samples showed low activity levels (0.21–0.41 nmol

product/minute/mg protein), and eight samples showed high activity levels ranging from 0.82 to 1.23 nmol/minute/mg protein. In another study of seven human subjects, lysates of erythrocytes showed high activities for producing formaldehyde from dichloromethane (presumably via GST-T1) in three subjects (15.4, 17.7, and 17.8 nmol product/minute/mg hemoglobin) and lower activity in the other four subjects (4.3, 6.0, 7.2, and 7.6 nmol product/minute/mg hemoglobin) (Hallier et al., 1994).

Comparisons of mice, rats, humans, and hamsters for the ability to metabolize dichloromethane via the GST pathway in liver and lung tissues indicate that mice appear to be the most active at metabolizing dichloromethane (Sherratt et al., 2002; Thier et al., 1998; Casanova et al., 1997, 1996; Hashmi et al., 1994; Reitz et al., 1989). Reitz et al. (1989) reported mean (± SD) GST enzymatic activity levels with dichloromethane as substrate (in units of nmol product formed/minute/mg protein) in liver cytosol preparations to be: 25.9 ± 4.2 units in $B6C3F_1$ mice (n = 15 determinations per preparation); 7.05 ± 1.7 units in F344 rats (n = 6); and 1.27 ± 0.21 units (n = 6) in Syrian golden hamsters. Mean GST activity levels in liver preparations from four human subjects (accident victims screened for human immunodeficiency virus and hepatitis B and C and obtained through a transplant center) were 2.62 ± 0.44 units (n = 10), –0.01 ± 0.04 units (n = 6), 2.71 ± 0.45 units (n = 6), and 3.03 ± 0.44 units (n = 6) (Reitz et al., 1989). The finding that one of the four individuals was unable to conjugate dichloromethane with GST was reflective of the estimated frequency of the GST-T1 null genotype in the U.S. population (see Table 3-3). Mean GST activity levels in lung cytosol preparations showed a similar rank order among species: 7.3 ± 1.4 units in mice (n = 4), 1.0 ± 0.1 units in rats (n = 4), 0.0 ± 0.2 units in hamsters (n = 4), and 0.37 ± 0.25 units in a pooled lung preparation from the same four human subjects (n = 2). Reitz et al. (1989) noted that relative abilities of these animal species to metabolize dichloromethane via the GST pathway correlated with their cancer sensitivities in long-term inhalation bioassays: (1) $B6C3F_1$ mice showed statistically significant increased incidence of liver and lung tumors in a 2-year cancer bioassay (National Toxicology Program [NTP], 1986); (2) rats showed much less evidence of increased incidence of liver tumors, and no increased risk of lung tumors at equivalent exposure concentrations but showed increased incidence of nonmalignant mammary tumors (NTP, 1986; Burek et al., 1984); and (3) Syrian golden hamsters did not show tumorigenic responses at any site (Burek et al., 1984).

Thier et al. (1998) conducted a study evaluating the activity of GST-T1 after treatment of dichloromethane in the cytosol of liver and kidney homogenates from hamsters (pooled male and females), rats (pooled male and female), male mice, and female mice and for humans classified as nonconjugators, low conjugators, or high conjugators of GST to dichloromethane. Little information is provided about the human samples other than that 13 kidney cancer patients were the source of the kidney samples; normal tissue identified by pathological exam was used. Blood samples from 10 of these patients were collected and enzyme activities measured in erythrocytes from 9 of these samples were reported. Results of conjugation of dichloromethane to GSH from

these studies are presented in Table 3-4. As can be seen from the table, activity levels (expressed as nmol/minute per mg of cytosolic protein) of humans varied considerably, with nonconjugators (presumed to be GST-T1$^{-/-}$) having no detectable activity, low conjugators (presumed to be GST-T1$^{+/-}$) having moderate activity, and high conjugators (presumed to be GST-T1$^{+/+}$) having approximately twice the activity seen in low conjugators. In the liver, the activity of rat GST conjugation was over twofold that seen in human high conjugators, while levels in mice were >11-fold (males) or 18-fold (females) greater than those of human high conjugators. In the kidney, the activity of high-conjugator humans was approximately 1.8-fold that of rats and was comparable to the activity of both male and female mice. The data in Table 3-4 show the following order for GST-T1 activities with dichloromethane as substrate: in liver preparations, mouse >> rat > human high conjugators > human low conjugators > hamster > human nonconjugators and, in kidney preparations, female mouse ≈ male mouse ≈ human high conjugators > rat ≈ human low > hamster > human nonconjugators. In addition, the data indicate that activity levels in liver, kidney, and erythrocytes of human subjects are in correspondence with the nonconjugator, low conjugator, and high conjugator designations.

Table 3-4. GST-T1 enzyme activities toward dichloromethane in human, rat, mouse, and hamster tissues (liver, kidney, and erythrocytes)

	Activity (nmol/min per mg protein)[a]		Activity (nmol/min per mL)[a]
	Liver	**Kidney**	**Erythrocytes**
Human, nonconjugators	Not detectable (2)	Not detectable (1)	Not detectable (1)
Human, low conjugators	0.62 ± 0.30 (11)	1.38 ± 0.52 (8)	9.67 ± 2.49 (5)
Human, high conjugators	1.60 ± 0.48 (12)	3.05 ± 0.72 (4)	18.28 ± 0.46 (3)
Rat	3.71 ± 0.28 (8)	1.71 ± 0.28 (8)	Not measured
Mouse, male	18.2 ± 2.22 (5)	3.19 ± 0.46 (5)	Not measured
Mouse, female	29.7 ± 6.31 (5)	3.88 ± 0.90 (5)	Not measured
Hamster	0.27 ± 0.20 (6)	0.25 ± 0.21 (6)	Not measured

[a]Mean ± SD with number of samples noted in parentheses.

Source: Adapted from Thier et al. (1998).

Sherratt et al. (2002) reported that, on a per mg basis, native recombinant mouse GST-T1 (purified after expression in *Escherichia coli*) was approximately twofold more active toward dichloromethane than native recombinant human enzyme, as well as being approximately fivefold more efficient (as assessed by the ratio of k_{cat}/K_m).

The distribution of GST-T1 in human tissues has been examined with antibodies raised against recombinant human GST-T1 (Sherratt et al., 2002, 1997). Immunoblotting of sodium dodecyl sulfate polyacrylamide gel electrophoresis gels loaded with tissue extracts from a 73 year-old man who had died with brochopneumonia and atherosclerosis indicated the

following order of expression of GST-T1: liver ≈ kidney > prostate ≈ small intestine > cerebrum ≈ pancreas ≈ skeletal muscle > lung ≈ spleen ≈ heart ≈ testis (Sherratt et al., 1997). It was estimated that the levels of cross-reacting materials in the cerebrum, pancreas, or skeletal muscle extracts were about 10% of those in the liver, whereas levels in the lung, spleen, heart, and testis were less than 5% of the levels in the liver. Comparison of the amounts of cross-reacting material in soluble liver extracts from a $B6C3F_1$ mouse and five human subjects (i.e., normal liver tissue samples from biopsies of secondary liver tumors) found that levels of GST-T1 protein were higher in the mouse extracts than in any of the human liver extracts (Sherratt et al., 2002). Densitometer analysis indicated that the GST-T1 level in the mouse liver extract was about fivefold higher than those in human liver extracts displaying the highest level. Cross-reacting material was not detectable in liver extracts from one of the five human subjects, indicating that this individual may have been GST-T1 null (Sherratt et al., 2002).

Results from in situ hybridization with oligonucleotide anti-sense probes for GST-T1 mRNA levels and immunohistochemical studies with antibodies to GST-T1 have indicated that there may be subtle differences between mice and humans in the intracellular localization of GST-T1 in the liver. Mainwaring et al. (1996) reported that staining for GST-T1 mRNA was higher in liver slices from $B6C3F_1$ mice than in liver slices from F344 rats and that staining in human liver samples was very low. Although the number of mouse and rat liver samples examined in this study was not indicated in the available report, it was reported that slices from five human liver samples were examined. No information was provided regarding the clinical history of the sources of the human samples. In mouse liver, staining for GST-T1 mRNA was enhanced in the limiting plate hepatocytes, in nuclei, in bile-duct epithelial cells, and in lesser amounts in the centrilobular cells in general. In rat liver, a similar pattern was observed, except no enhanced staining was observed in the limiting plate hepatocytes or in nuclei. Staining for GST-T1 mRNA in the human liver samples showed an even distribution throughout the liver lobule, and no mention of a specific nuclear localization was made (Mainwaring et al., 1996). Quondamatteo et al. (1998), using antibodies to GST-T1, subsequently reported a similar localization of GST-T1 protein in nuclei of cells in mouse liver slices. In another study using antibodies raised against recombinant human GST-T1 or a peptide derived from the deduced mouse GST-T1 primary sequence, Sherratt et al. (2002) reported that nuclear staining was observed in all cells in mouse liver slices (from five individual $B6C3F_1$ mice) showing the presence of mouse GST-T1; staining in the cytoplasm was only detected in cells with very high levels of GST-T1. In liver slices obtained from two human subjects (males, ages 60 and 61 years, with a secondary liver tumor and what was described as a "cavernous hemangioma" without malignancy, respectively), the most intense nuclear staining was associated with bile duct epithelial cells, but there was heterogeneity of staining within hepatocytes; some cells showed nuclear staining, but others only exhibited cytoplasmic staining (Sherratt et al., 2002).

In summary, the relative amount of dichloromethane metabolized via the GST pathway increases with increasing exposure concentrations. As the high affinity CYP pathway becomes saturated (either from high exposure levels of genetic or other factors that decrease CYP2E1 activity), the GST pathway increases in relative importance as a dispositional pathway for dichloromethane. Two reactive metabolites (S-(chloromethyl)glutathione and formaldehyde) resulting from this pathway have been identified. GST-T1 is the GST isozyme that catalyzes conjugation of dichloromethane with GST. Interindividual variation in the ability to metabolize dichloromethane via GST-T1 is associated with genetic polymorphisms in humans. Estimated U.S. population prevalence of nonconjugators (–/– at the GST-T1 locus) is about 20%, but higher prevalences (47–64%) have been reported for Asians (Raimondi et al., 2006; Haber et al., 2002; Garte et al., 2001; Nelson et al., 1995). The prevalences for low (+/– at the GST-T1 locus) and high (+/+) conjugators have been estimated at 48 and 32%, respectively (Haber et al., 2002). The liver and kidney are the most enriched tissues in GST-T1, but evidence is available for the presence of GST-T1 in other tissues at lower levels, including the brain and lung. In humans, GST-T1 expression in the brain is lower than that seen in the liver or kidney but higher than in the lung. Comparisons of mice, rats, humans, and hamsters for the ability to metabolize dichloromethane via the GST pathway in liver (based on measurement of tissue-specific enzyme activity) indicate the following rank order: mice > rats > or ≈ humans > hamsters. This relative ranking corresponds to the rank order of the strength of the association between inhalation exposure to dichloromethane and liver tumors in long-term cancer bioassays with mice, rats, and hamsters. In mouse liver tissue, GST-T1 appears to be localized in the nuclei of hepatocytes and bile-duct epithelium, but rat liver does not show preferential nuclear localization of GST-T1. In human liver tissue, some hepatocytes show nuclear localization of GST-T1 and others show localization in cytoplasm, as well as in bile duct epithelial cells. The apparent species differences in intracellular localization of GST-T1 may play a role in species differences in susceptibility to dichloromethane carcinogenicity if nuclear production of S-(chloromethyl)glutathione is more likely to lead to DNA alkylation than cytoplasmic production.

3.4. ELIMINATION

Dichloromethane is eliminated mainly through exhalation either of the parent compound or as the two primary metabolites CO_2 and CO (Angelo et al., 1986a, b; McKenna et al., 1982; DiVincenzo and Kaplan, 1981; DiVincenzo et al., 1972, 1971). In human studies, dichloromethane is rapidly eliminated from the body following the cessation of exposure, with much of the parent compound completely removed from the bloodstream and expired air by 5 hours postexposure in experiments using exposure levels of 90, 100, or 210 ppm (DiVincenzo et al., 1972, 1971; Riley et al., 1966). Studies in rats have similarly demonstrated that elimination from the blood is rapid, with elimination half-times in F344 rats on the order of 4–

6 minutes following intravenous doses in the range of 10–50 mg/kg (Angelo et al., 1986a). In a study using Sprague-Dawley rats, Carlsson and Hultengren (1975) demonstrated variability in elimination rates between different types of tissues, with the most rapid elimination seen in the adipose and brain tissue, while elimination from liver, kidneys, and adrenals proceeded more slowly.

In a study using human volunteers, DiVincenzo and Kaplan (1981) reported a dose-related increase in CO in the expired breath after inhalation exposure to 50–200 ppm of dichloromethane, with a net elimination as CO on the order of 25–35% of the absorbed dose. Similar results have been reported in animal studies. Following gavage administration of 50 or 200 mg/kg-day doses of [^{14}C]-labeled dichloromethane in water to groups of six mature male F344 rats for up to 14 days, >90% of the label was recovered in the expired air within 24 hours of dose administration (Angelo et al., 1986b). Following administration of the first of 14 daily 50 mg/kg-day doses, radioactivity in parent compound, CO_2, and CO in the 24-hour expired breath accounted for 66, 17, and 16% of the administered radioactivity, respectively; similar patterns were reported for 24-hour periods following administration of the seventh and fourteenth 50 mg/kg-day dose. Following administration of the first 200 mg/kg-day dose, radioactivity in parent compound, CO_2, and CO in the 24-hour expired breath accounted for 77, 9, and 6%, respectively, of the administered radioactivity (Angelo et al., 1986b). In mature, male Sprague-Dawley rats given a smaller dose (1 mg/kg) of [^{14}C]-labeled dichloromethane, radioactivity in parent compound, CO_2, and CO in 48-hour expired breath accounted for 12, 35, and 31%, respectively; these data indicate that, at lower dose levels, a greater percentage of the administered dose was metabolized by the CYP pathway and eliminated in the expired breath, compared with higher dose levels (McKenna and Zempel, 1981). Similar patterns of radioactivity distribution in parent compound, CO_2, and CO in expired breath were found in mature male B6C3F$_1$ mice following gavage administration of 50 mg/kg-day (in water), or 500 or 1,000 mg/kg-day (in corn oil), [^{14}C]-labeled dichloromethane (Angelo et al., 1986a). For example, radioactivity in parent compound, CO_2, and CO in 24-hour expired breath accounted for 61, 18, and 11% of the administered radioactivity, following administration of a single 50 mg/kg dose to a group of six mice (Angelo et al., 1986a). Exhalation rates were similarly high following inhalation exposure of mature male Sprague-Dawley rats (>90%) (McKenna et al., 1982) or following intravenous administration of dichloromethane to mature male F344 rats (Angelo et al., 1986b).

Elimination of dichloromethane in the urine of exposed humans is generally small, with total urinary dichloromethane levels on the order of 20–25 μg or 65–100 μg in 24 hours following a 2-hour inhalation exposure to 100 or 200 ppm, respectively (DiVincenzo et al., 1972). However, a direct correlation between urinary dichloromethane and dichloromethane exposure levels was found in volunteers, despite the comparatively small urinary elimination (Sakai et al., 2002). Following administration of a labeled dose in animals, regardless of

exposure route, generally <5–8% of the label is found in the urine and <2% in the feces (McKenna et al., 1982; McKenna and Zempel, 1981; DiVincenzo et al., 1972, 1971).

3.5. PHYSIOLOGICALLY BASED TOXICOKINETIC MODELS

Several PBTK models for dichloromethane in animals and humans have been developed from 1986 to 2006. These models are mathematical representations of the body and its absorption, distribution, metabolism, and elimination of dichloromethane and select metabolites, based on the structure of the Ramsey and Andersen (1984) model for styrene. The models' equations are designed to mimic actual biological behavior of dichloromethane, incorporating in vitro and in vivo data to define physiological and metabolic equation parameters. As such, the models can simulate animal or human dichloromethane exposures and predict a variety of dichloromethane and metabolite internal dosimeters (i.e., instantaneous blood and tissue concentration, area under the curve [AUC] of concentration versus time plots, rate of metabolite formation), allowing for the extrapolation of toxicity data across species, route of exposure, and high to low exposure levels. The development of dichloromethane PBTK models has resulted in either increased biological detail and functionality or refinement of model parameters with newly available data. The former type of development provides more options for toxicity data extrapolation, while the latter serves to increase confidence in model predictions and decrease uncertainty in risk assessments for which the models were, or will be, applied. This section of the document describes each of the models reported in the scientific literature and/or used by the regulatory community (i.e., Occupational Safety and Health Administration [OSHA], EPA) and their contribution to the advancement of predictive dosimetry and data extrapolation for dichloromethane. In some instances, model development was accomplished by the addition of new biological compartments (e.g., tissue systems). Diagrams of the compartmental structure of the models are shown in Figure 3-2. Significant statistical advances in parameter estimation also have been incorporated in model development. For this reason, some animal and human PBTK models may be described as deterministic (Sweeney et al., 2004; Casanova et al., 1996; Reitz et al., 1988a, b; U.S. EPA, 1988b, 1987a, b; Andersen et al., 1987; Gargas et al., 1986) in which point estimates for each model parameter are used, resulting in point estimates for dosimetry. Others may be described as probabilistic (Jonsson and Johanson, 2001; El-Masri et al., 1999; OSHA, 1997), in which probability distributions for each parameter were defined, resulting in probability distributions for dosimetry. The latter approach, particularly utilizing a Bayesian hierarchical statistical model structure (described below) (David et al., 2006; Marino et al., 2006) to estimate parameter values, allows for the introduction of intra- and interspecies variability into model predictions and quantitative assessment of model uncertainty. Both deterministic (U.S. EPA, 1988b, 1987a, b) and probabilistic (OSHA, 1997) applications have been used to develop regulatory values. As discussed below, subsequent applications of the developed models for cancer risk assessment have resulted in significantly different estimates of human cancer risk.

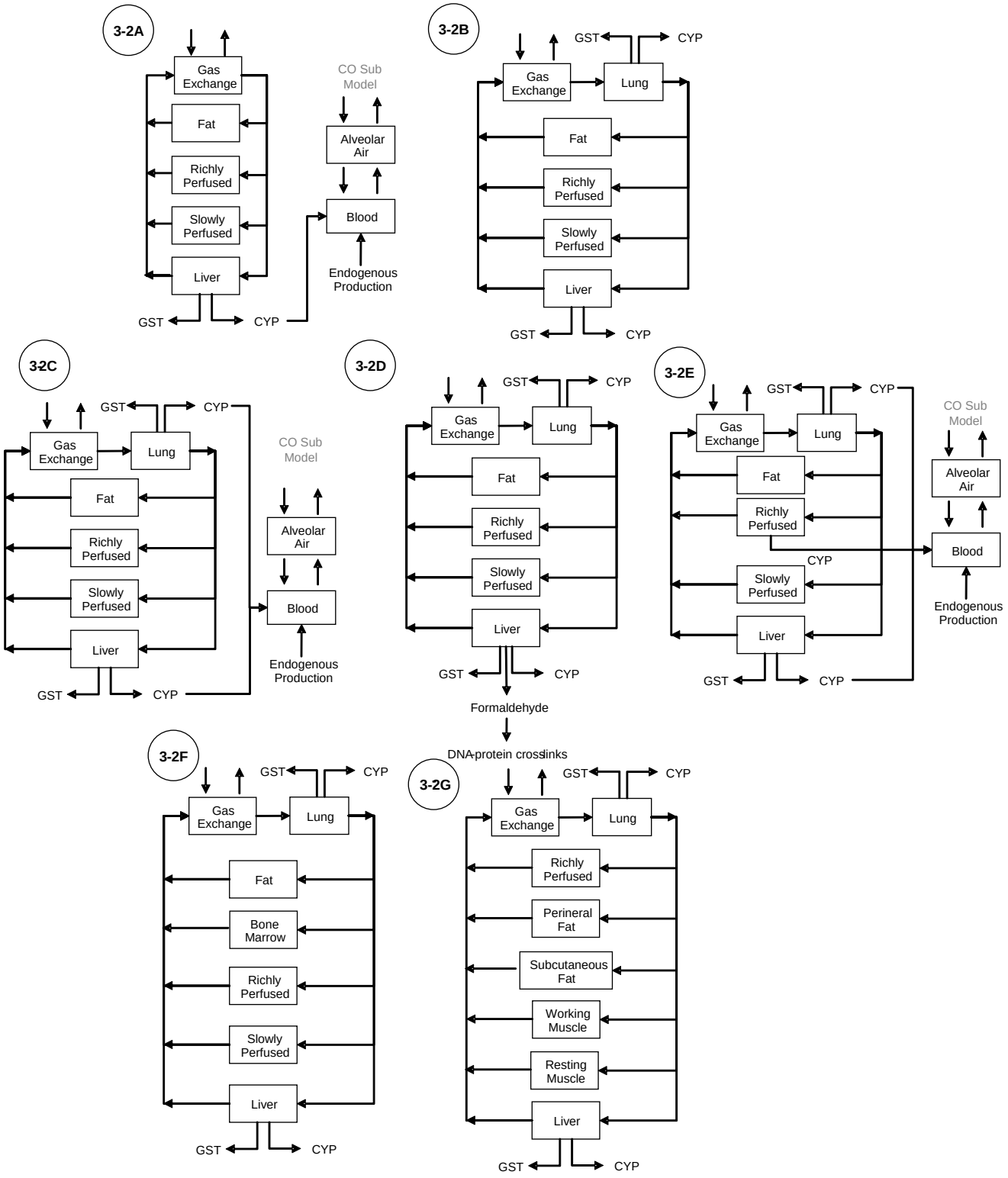


Figure 3-2. Schematics of PBTK models (1986–2006) used in the development of estimates for dichloromethane internal dosimetry.

Key references: Model A—Gargas et al. (1986); B—Andersen et al. (1987); C—Andersen et al. (1991); D—Casanova et al. (1996); E—Sweeney et al. (2004); F—OSHA (1997); G—Jonsson and Johanson (2001). Models C–G all build on the structure in model B. Models E and G have been applied in humans; all others have been applied in humans and rodents (mice and/or rats). CYP = CYP pathway metabolites; GST = GST pathway metabolites.

The deterministic rat model of Gargas et al. (1986), based on previous work by Ramsey and Andersen (1984) examining inhalation pharmacokinetics of styrene in rats, was the first PBTK model for dichloromethane. It was comprised of four compartments (fat, liver, richly perfused tissues, and slowly perfused tissues [Figure 3-2A]) and described flows and partitioning of parent material and metabolites through the compartments with differential equations. Metabolism, which was restricted to the liver compartment, was described as two competing pathways: the GST pathway, described with a linear first-order kinetic model, and the CYP pathway, described with a saturable Michaelis-Menten kinetic model. Rate constants for the CYP and GST pathways in rats were determined by optimization of the model with in vivo gas uptake data. COHb production was modeled both endogenously and from CYP-mediated metabolism of dichloromethane. This model demonstrated the dose-dependent flux through the competing CYP and GST metabolic pathways and the effect of CYP inhibition on COHb generation.

Andersen et al. (1987) extended the rat model of Gargas et al. (1986) to include a lung compartment, including CYP and GST metabolism pathways within the lung, in rats, mice, hamsters, and humans (Figure 3-2B). Physiological flow rates were allometrically scaled among species by ¾ power of body weight (BW). Rate constants for the CYP and GST pathways in rodents were determined by optimization of the model with in vivo gas uptake data. CYP rate constants for humans were derived from data on dichloromethane uptake in human subjects (number of subjects not reported). Human GST rate constants were derived by allometric scaling of the animal GST rate constants. Model predictions compared favorably with kinetic data for human subjects exposed by inhalation to dichloromethane (Andersen et al., 1987). Using the mouse cancer bioassay data from NTP (1986), Andersen et al. (1987) compared the linear body surface area-derived or the PBTK model-derived human liver and lung dose surrogates associated with tumor development (mg dichloromethane metabolized via GST pathway/volume tissue/day). They reported that PBTK model-extrapolated human liver and lung internal doses were for inhalation exposure 167- and 144-fold lower and for drinking water exposure 45- and 213-fold lower, respectively, than body surface area scaled internal doses. The study authors suggested that the lower model-predicted human internal dose surrogates were due to the need to saturate the CYP pathway before appreciable tumorigenic metabolite levels could be attained, which is not captured by extrapolation based on body surface area.

U.S. EPA (1988b, 1987a, b) slightly modified the Andersen et al. (1987) model for mice by using different alveolar ventilation and cardiac flow rates and used the mouse and human models to derive human cancer risks from animal tumor incidence data. The flow rate parameters in the Andersen et al. (1987) model were based on a human breathing rate of 12.5 m^3/day (reflecting a resting rate), compared with the EPA value of 20 m^3/day (reflecting average daily activity level) and a mouse breathing rate of 0.084 m^3/day (based on allometric scaling of bioassay-specific BWs), compared with the rate commonly used by EPA,

0.043 m^3/day (U.S. EPA, 1987a). The internal dose metric used in the applications of the model to cancer risk assessment was reflective of the amount of dichloromethane metabolized by the GST pathway. In addition to using the mouse and human PBTK models to account for species differences in dosimetry, a body surface area correction factor of 12.7 was applied to low-dose slopes of estimated dose-response relationships for liver and lung tumors in mice to account for presumed higher human responsiveness, relative to mice, to dichloromethane-induced cancer (U.S. EPA, 1987a). The factor of 12.7 is the cube root of the ratio of human to mouse reference BWs; this BW scaling factor was applied to adjust for interspecies toxicodynamic variability (i.e., presumed differences in the lifetime impact in mice and humans of a given daily amount of dichloromethane metabolically activated per liter of tissue) (Rhomberg, 1995). A human cancer inhalation unit risk (IUR) of 4.7×10^{-7} per ($\mu g/m^3$), based on this analysis, was placed on IRIS in September 1990.

The Andersen et al. (1987) models were also modified by addition of submodel structures for estimation of new dosimeters of interest. Andersen et al. (1991) added the capability to specifically describe the kinetics of dichloromethane, CO, and COHb in rats and humans with the addition of the Coburn-Forster-Kane equation to describe CO and COHb kinetics (Figure 3-2C). However, equations were not added for metabolism of dichloromethane to CO in the lung. Casanova et al. (1996) extended the Andersen et al. (1987) mouse model to include a submodel that predicted the formation of formaldehyde and DNA-protein cross-links in the liver (Figure 3-2D).

Further refinements of the Andersen et al. (1987) models allowed for incorporation of new data. New in vitro measurements of metabolic rate constants in human and animal tissues were incorporated into the Andersen et al. (1987) models by Reitz and coworkers (Reitz, 1991; Reitz et al.1988a, b). Sweeney et al. (2004) modified the Andersen et al. (1987) human PBTK model, adding extrahepatic CYP metabolism in richly perfused tissues (Figure 3-2E) to obtain a better fit of the model to kinetics data for humans. Data for 13 volunteers (10 men and 3 women) who were exposed to one or more concentrations of dichloromethane for 7.5 hours included dichloromethane concentrations in breath and blood, COHb concentrations in blood, and CO concentrations in exhaled breath. Individual CYP $V_{max}c$ (maximal velocity) values were obtained by optimizing model predictions to match time-course data simultaneously for dichloromethane concentrations in blood and exhaled breath for each individual. Resultant individual values of CYP $V_{max}c$ ranged from 7.4 to 23.6 mg/hour/$kg^{0.7}$, indicating an approximate threefold range in maximal CYP metabolic activity.

The significance of metabolic variability for the kinetics of dichloromethane in animals and humans was explored by several investigators using PBTK models. Dankovic and Bailer (1994) used the updated human model presented by Reitz et al. (1988a, b) to explore the consequences of interindividual variability in vitro kinetic constants for the CYP and GST pathways (based on data for four human subjects) and reported that predicted GST-metabolized

doses to the lung and liver could range from about zero to up to fivefold greater than those predicted with the values of these rate constants used in the Reitz et al. (1988a, b) model. El-Masri et al. (1999) replaced parameter estimates in the mouse and human PBTK models presented by Casanova et al. (1996) with probability distributions, including published information on the distribution of GST-T1 polymorphism in human populations, and used Monte Carlo simulations to estimate distributions of cancer potency of dichloromethane in mice, distributions of the amount of DNA-protein cross-links formed in the liver of humans, and distributions of human cancer risks at given exposure levels of dichloromethane. The analysis showed that, at exposure levels of 1, 10, 100, and 1,000 ppm dichloromethane, average and median cancer risk estimates were 23–30% higher when GST-T1 polymorphism was not included in the model.

Given the demonstrated influence of population variability in dichloromethane metabolism on PBTK model-derived cancer risk estimates (El-Masri et al., 1999; Dankovic and Bailer, 1994), PBTK model development has included a more formal statistical treatment of data for physiological and metabolic variability. Bayesian statistical approaches have been applied to develop probabilistic PBTK models for dichloromethane. Probabilistic models account for variability between individuals in model parameters by replacing point estimates for the model parameters with probability distributions. Calibration or fitting of probabilistic PBTK models to experimental toxicokinetic data is facilitated by a Bayesian technique called Markov Chain Monte Carlo (MCMC) simulation, which quantitatively addresses both variability and uncertainty in PBTK modeling (Jonsson and Johanson, 2003).

OSHA (1997) used MCMC simulation to fit probabilistic versions of the Reitz et al. (1988a, b) and Andersen et al. (1991, 1987) mouse and human models, which included probability distributions for all model parameters. GST- and CYP-mediated metabolism occurred in the liver and lung compartments (see Figure 3-2F). The model parameters were modified to focus on occupational exposure scenarios; that is, a parameter distribution for work intensity (using data from Astrand [1989]) was added, which adjusted physiological flow rates as a function of work intensity as measured in watts. In addition, updated measurements of blood:air and tissue:air partition coefficients (Clewell et al., 1993) were used to describe distributions for these parameters. The Clewell et al. (1993) blood:air partition coefficient of 23 is higher than the value of 8.29 reported by Andersen et al. (1987) and used by EPA (U.S. EPA, 1988b, 1987a, b). The newer Clewell et al. (1993) value for mice is the preferred value, since it is much closer to the values for rats (19.4) and hamsters (22.5) rather than humans (9.7), as reported by Andersen et al. (1987). Distributions of metabolic, physiological, and partitioning parameters in the mouse and human models were updated by using Bayesian methods with data for mice and humans in published studies of mouse and human physiology and dichloromethane kinetic behavior.

Jonsson et al. (2001) used additional human kinetics data to expand the PBTK model of Reitz et al. (1988a, b) and added new model compartments (Figure 3-2G). These investigators used MCMC simulation to develop a probabilistic model from the Reitz et al. (1988a, b) human model by using published in vitro measurements of liver V_{max} for the CYP pathway (Reitz et al., 1989) and kinetic data for five human subjects exposed by inhalation to dichloromethane (Astrand et al., 1975). A working muscle compartment was added to the basic Andersen et al. (1987) and Reitz et al. (1988a, b) structure (see Figure 3-2G). Jonsson and Johanson (2001) refined and extended this probabilistic model by including an additional fat compartment (to provide a better description of the experimental data for the time course of dichloromethane in subcutaneous fat), incorporating (with MCMC simulation) kinetic data for dichloromethane in an additional 21 human subjects and including three GST-T1 genotypes/phenotypes (nonconjugators –/–, low conjugators +/–, high conjugators +/+). Monte Carlo simulations were then used with the refined probabilistic model to predict human liver cancer risk estimates at several dichloromethane exposure levels using an algorithm similar to the one used by El-Masri et al. (1999), using DNA-protein cross-links as the internal dose metric. The mean, 50^{th}, 90^{th}, and 95^{th} percentile human cancer risk values from Jonsson et al. (2001) and El-Masri et al. (1999) were very similar, within onefold of one another for simulated exposure levels up to 100 ppm.

The most statistically rigorous and data-intensive PBTK model development was performed by Marino et al. (2006) for mice and David et al. (2006) for humans. Development of these models used multiple mouse and human data sets in a Bayesian hierarchical statistical structure to quantitatively capture population variability and reduce uncertainty in model dosimetry and the resulting risk values. EPA used these models in the derivation of reference values and cancer risk estimates in the current assessment, and these models are described in more detail below.

3.5.1. Probabilistic Mouse PBTK Dichloromethane Model (Marino et al., 2006)

Marino et al. (2006) used MCMC analysis to develop a probabilistic PBTK model for dichloromethane in mice, using the Andersen et al. (1987) model structure as a starting point (Figure 3-3). Metabolic kinetic parameters ($V_{max}c$, K_m, k_fC, A1, and A2) (Table 3-5) were calibrated with this Bayesian methodology by using several experimental data sets. Distribution parameters (i.e., means and coefficients of variation [CVs]) for other physiological parameters (i.e., BW, fractional flow rates, and fractional tissue volumes) and partition coefficients were taken from the general literature as noted by Clewell et al. (1993). Marino et al. (2006) noted that using distributions for these latter parameters from the general literature (based on a large number of animals) was better than updating them based on the relatively smaller number of animals in the available dichloromethane kinetic studies. Clewell et al. (1993) determined blood:air and tissue:air partition coefficients (means and CVs) with tissues from groups of male

and female B6C3F$_1$ mice. These partition coefficients were derived by using a vial equilibration method similar to that used by prior investigators (Andersen et al., 1987; Gargas et al., 1986). Tissue:air partition coefficients were approximately two to three times lower than previously utilized values with the exception of the liver coefficient, which was similar to previous values (Table 3-5). The blood:air partition coefficient (23) from Clewell et al. (1993) is higher than the previously reported value of 8.3 (Gargas et al., 1986). The higher value is more in line with values measured in rats (19.4) and hamsters (22.5) and, thus, is more reasonable than the older value of 8.3. Table 3-5 shows mean and CVs for physiological parameters and partition coefficients in the Marino et al. (2006) mouse model as well as values used in earlier deterministic PBTK mouse models for dichloromethane.

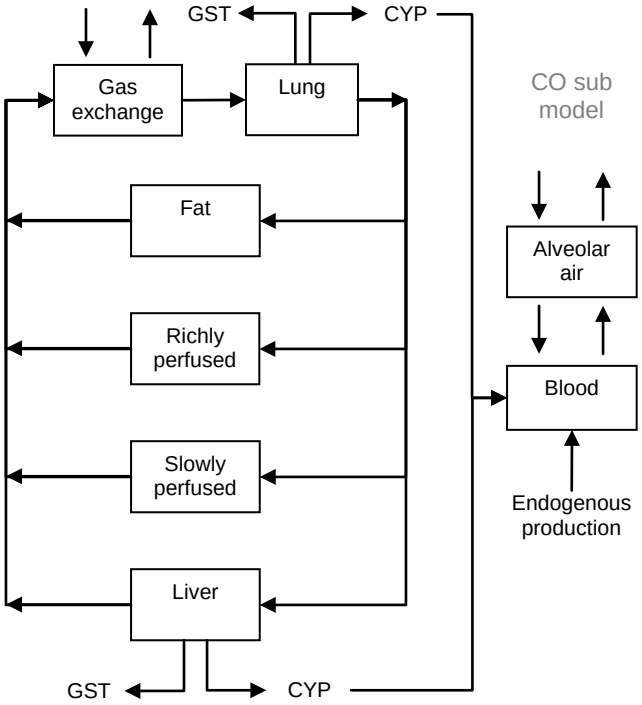


Figure 3-3. Schematic of mouse PBTK model used by Marino et al. (2006).

The Bayesian calibration of the cardiac output constant (QCC), ventilation:perfusion ratio (VPR), and metabolic parameters was divided into three sequential steps: using kinetic data from closed chamber studies with mice treated with an inhibitor of CYP2E1 (trans-1,2-dichloroethylene) in order to minimize the oxidative pathway and enable a more precise estimate of parameters for the GST pathway, followed by kinetic data for mice given intravenous injections of dichloromethane to estimate metabolism parameters in the absence of pulmonary absorption processes and, finally, kinetic data for naïve mice exposed to dichloromethane in closed chambers (Marino et al., 2006). The initial prior distributions were based on mean values

used by Andersen et al. (1987) for the metabolic parameters and by OSHA (1997) for the parameters for VPR, ratio of lung V_{max} to liver V_{max} (A1), and ratio of lung GST 1st order kinetic constant (lung K_F) to liver K_F (A2). Posterior distributions from the first Bayesian analysis were used as prior distributions for the second step, and posterior distributions from the second step were used as prior distributions for the final updating. Final results from the Bayesian calibration of the mouse probabilistic model are shown in Table 3-5.

Table 3-5. Values for parameter distributions in a B6C3F$_1$ mouse probabilistic PBTK model for dichloromethane compared with associated values for point parameters in earlier deterministic B6C3F$_1$ mouse PBTK models for dichloromethane

Parameter	Prior mean	Prior CV	Marino et al. (2006)[a] Final posterior mean	Marino et al. (2006)[a] Final posterior CV	U.S. EPA (1988b, 1987a, b)	Andersen et al. (1987)
Fractional flow rates (fraction of QCC)[b]						
QFC Fat	0.05	0.60			0.05	0.05
QLC Liver	0.24	0.96			0.24	0.24
QRC Rapidly perfused tissues	0.52	0.50			0.52	0.52
QSC Slowly perfused tissues	0.19	0.40			0.19	0.19
Fractional tissue volumes (fraction of BW)[b]						
VFC Fat	0.04	0.30	These parameters were taken from an		0.04	0.04
VLC Liver	0.04	0.06	extensive literature database derived from		0.04	0.04
VLuC Lung	0.0115	0.27	a large number of animals; therefore,		0.0119	0.0119
VRC Rapidly perfused tissues	0.05	0.30	further Bayesian updating does not		0.05	0.05
VSC Slowly perfused tissues	0.78	0.30	inform on the true mean and variance for		0.78	0.78
Partition coefficients[c]			these values.			
PB Blood:air	23	0.15			8.29	8.29
PF Fat:blood	5.1	0.30			14.5	14.5
PL Liver:blood	1.6	0.20			1.71	1.71
PLu Lung:blood	0.46	0.27			1.71	1.71
PR Rapidly perfused:blood	0.52	0.20			1.71	1.71
PS Slowly perfused:blood	0.44	0.20			0.96	0.96
Flow rates						
QCC Cardiac output (L/hr/kg$^{0.74}$)	28.0	0.58	24.2	0.19	14.3[d]	28.0[e]
VPR ventilation:perfusion ratio	1.52	0.75	1.45	0.20	1.0	1.0
Metabolism parameters						
$V_{max}c$ Maximum CYP metabolic rate (mg/hr/kg$^{0.7}$)	11.1	2	9.27	0.21	11.1	11.1
K_m CYP affinity (mg/L)	0.396	2	0.574	0.42	0.396	0.396
k_fC First-order GST metabolic rate constant (kg$^{0.3}$/hr)	1.46	2	1.41	0.28	1.46	1.46
A1 Ratio of lung $V_{max}c$ to liver $V_{max}c$	0.462	0.55	0.207	0.36	0.416	0.416
A2 Ratio of lung k_fC to liver k_fC	0.322	0.55	0.196	0.37	0.137	0.137

[a]MCMC analysis was used to update prior distributions (means and CVs) for flow rate and metabolic parameters in a sequential process with three sets of kinetic data from mouse studies, as explained further in the text. Final values for posterior distributions are given in this table.
[b]Source: Andersen et al. (1987, 1991).
[c]Source: Clewell et al. (1993).
[d]Based on a mouse breathing rate of 0.043 m^3/day.
[e]Based on a mouse breathing rate of 0.084 m^3/day.

Marino et al. (2006) used the Bayesian-calibrated mouse model to calculate internal dose metrics associated with exposure conditions in the NTP (1986) $B6C3F_1$ mouse cancer inhalation bioassay. The internal dose metric selected was milligrams (mg) dichloromethane metabolized by the GST pathway per liter tissue per day. This is the same dose metric used in earlier applications of PBTK models to derive human cancer IUR estimates based on cancer responses in mice (OSHA, 1997; Andersen et al., 1987; U.S. EPA, 1987a, b). Its use is consistent with evidence that dichloromethane metabolism via GST-T1 results in the formation of a reactive metabolite that damages DNA and results in the formation of tumors (see section 4.7). The model was used to calculate values for this internal dose metric in the lung and liver of mice in the NTP (1986) study, using the mean values of the final distributions for the parameters in the model. Resultant values were three- to four-fold higher than values calculated with the Andersen et al. (1987) and U.S. EPA (1987a, b) versions of the model (Table 3-6). Marino et al. (2006) noted that the difference could be primarily attributed to the changes in the partition coefficients based on Clewell et al. (1993) as well as to the Bayesian updating of the metabolic parameters (see Table 3-5).

Table 3-6. Internal daily doses for $B6C3F_1$ mice exposed to dichloromethane for 2 years (6 hours/day, 5 days/week) calculated with different PBTK models

Target organ	NTP (1986) exposure level[a]	PBTK model		
		Marino et al. (2006)	U.S. EPA (1987a, b)	Andersen et al. (1987)
Liver[b]	Control	0	0	0
	2,000 ppm	2,359.99	727.8	851
	4,000 ppm	4,869.85	1,670	1,811
Lung[b]	Control	0	0	0
	2,000 ppm	474.991	111.4	123
	4,000 ppm	973.343	243.7	256

[a]2,000 ppm = 6,947 mg/m^3; 4,000 ppm = 13,894 mg/m^3.
[b]Internal dose expressed as mg dichloromethane metabolized by the GST pathway per liter tissue per day.

Marino et al. (2006) noted that inclusion of extrahepatic CYP metabolism in the slowly perfused tissue compartment in the mouse model had little impact on the formation of GST metabolites in the liver and lung, especially at exposure levels used in the mouse NTP (1986) bioassay. To support this contention, the Andersen et al. (1987) model was modified to include 10% of the liver rate of oxidative metabolism in the slowly perfused tissue compartment (as suggested by Sweeney et al. [2004]), and the modified model was used to calculate the formation of GST metabolites. If extrahepatic metabolism was included in the slowly perfused tissue compartment, there was a 5–6% reduction in the formation of GST metabolites in the lung and liver at an exposure level of 50 ppm. At 2,000 or 4,000 ppm, however, there was only a 0.77 or 0.37% reduction, respectively. Marino et al. (2006) did not discuss the impact of including

extrahepatic metabolism in the rapidly perfused tissue compartment; the same group of investigators developed a human PBTK model that included CYP metabolism in the richly perfused compartment (David et al., 2006).

3.5.2. Probabilistic Human PBTK Dichloromethane Model (David et al., 2006)

The basic model structure used by David et al. (2006) was that of Andersen et al. (1987) with the addition of the CO submodel of Andersen et al. (1991), refinements from the Marino et al. (2006) mouse model, and an inclusion of CYP metabolism in richly perfused tissue (Figure 3-4). David et al. (2006) used Bayesian analysis to develop and calibrate metabolic parameters in a human probabilistic PBTK model for dichloromethane, using kinetic data from several studies of volunteers exposed to dichloromethane (n = 13 from DiVincenzo and Kaplan [1981]; n = 12 from Engström and Bjurström [1977]; n = 14 from Astrand et al. [1975]; n = 3 from Stewart et al. [1972a], and group means for metabolism parameters from Andersen et al. [1991]). Exhaled dichloromethane and CO and blood levels of dichloromethane and COHb were available in the studies by Andersen et al. (1991) and DiVincenzo and Kaplan (1981). The other three studies included two or three of these measures. The only available data for levels of dichloromethane in fat came from the study of Engström and Bjurström (1977) (described in section 3.2 within adipose tissue).

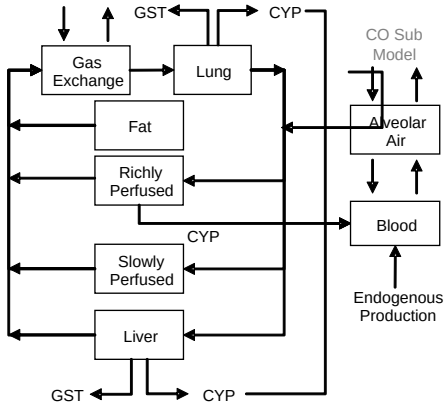


Figure 3-4. Schematic of human PBTK, used by David et al. (2006).

Values (means and SDs or CVs) for the model parameter distributions were selected from multiple sources considered to provide the most current scientific evidence for each parameter (David et al., 2006). Mean values for cardiac output (QCC), VPR, and all fractional tissue volumes and blood flow rates were based on mean values used by EPA (U.S. EPA, 2000d) in a PBTK model for vinyl chloride, as were values for CVs for all physiological parameters, except CVs for VPR and fractional lung volume, which were set to those used by OSHA (1997). Means

for the CO submodel parameters were set equal to those in Andersen et al. (1991), except for those for the endogenous rate of CO production (REnCOC) and the background amount of CO (ABCOC), which were based on data collected by DiVincenzo and Kaplan (1981). Means for partition coefficients, the ratio of lung V_{max} to liver V_{max}, and the ratio of lung K_F to liver K_F (A2) were those used by Andersen et al. (1987), whereas prior means for $V_{max}c$ and K_m were those used by Andersen et al. (1991). The prior mean for the metabolic parameter for CYP metabolism in the rapidly perfused tissue was set at 0.03, slightly lower than the value suggested by Sweeney et al. (2004). Prior CVs for the metabolic parameters were set at 200%.

MCMC analysis was used to calibrate metabolic parameters in the human model in a two-step approach: (1) posterior distributions were estimated separately by using data from each of the five studies with kinetic data for humans exposed to dichloromethane (with durations ranging from 1 to 8 hours and concentrations ranging from 50 to 1,000 ppm); and (2) posterior distributions were estimated with combined data from the 42 individual subjects from the four studies with individual subject data (DiVincenzo and Kaplan, 1981; Engström and Bjurström, 1977; Astrand et al., 1975; Stewart et al., 1972a). Results from the Bayesian calibration with the combined kinetic data for individual subjects are shown in Table 3-7. This analysis resulted in a narrowing of the distribution for the CYP2E1 metabolism parameters V_{max} and K_m, from a fairly broad prior distribution with a CV of 200% for both parameters to 13.1 and 33.6%, respectively, for V_{max} and K_m.

Table 3-7. Results of calibrating metabolic parameters in a human probabilistic PBTK model for dichloromethane with individual kinetic data for 42 exposed volunteers and MCMC analysis

	Prior distributions		Posterior distributions	
Parameter	**Mean (arithmetic)**	**CV**	**Mean (arithmetic)**	**CV**
$V_{max}c$ —-maximal CYP metabolic rate (mg/hr/kg$^{0.7}$)	6.25	2	9.42	0.131
K_m—CYP affinity (mg/L)	0.75	2	0.433	0.336
k_fC—first-order GST metabolic rate (kg$^{0.3}$/hr)	2	2	0.852	0.711
A1—ratio of lung $V_{max}c$ to liver $V_{max}c$	0.00143	2	0.000993	0.399
A2—ratio of lung k_fC to liver k_fC	0.0473	2	0.0102	0.728
FracR—fraction of $V_{max}c$ in rapidly perfused tissues	0.03	2	0.0193	0.786

Source: David et al. (2006).

The parameter statistics shown in Table 3-7 (values reported by David et al., 2006) are summary statistics of the converged parameter chains obtained in that analysis, leaving out any evaluation of correlation or covariance among the updated parameters. As such, these statistics implicitly include both the inter-individual variability that would have been elucidated by the

Bayesian analysis (variation between mean values for each individual for which data were available) and uncertainty in those values.

David et al. (2006) further refined the human probabilistic model to reflect polymorphisms in the GST pathway: homozygous positive (+/+) GST-T1 individuals, heterozygous (+/–) GST-T1, and homozygous negative (–/–) GST-T1 individuals with no GST activity. Distributions of GST activities for these genotypes in a group of 208 healthy male and female subjects from Sweden were scaled to obtain distributions of k_fC for each genotype that, when weighted by estimated frequencies of the genotypes in the U.S. population, would result in an overall population mean equal to the k_fC mean for the posterior distribution shown in Table 3-7 (0.852 $kg^{0.3}$/hour). The resultant mean k_fC values were 0.676 $kg^{0.3}$/hour (SD 0.123) for heterozygous individuals and 1.31 $kg^{0.3}$/hour (SD 0.167) for homozygous positive individuals. The final parameter distributions used by David et al. (2006) are summarized in Table 3-8.

As described in Appendix B, EPA undertook an evaluation of the David et al. (2006) model and parameterization, focusing on the adequacy of the characterization of parameter distributions in the full human population. EPA's conclusion is that the reported distributions for physiological parameters in particular, but also key metabolic parameters, only represented a narrow set of adults (with the exception of BW). The EPA therefore chose to use supplemental data sources to define these distributions in a way that should fully characterize the variability in the human population for individuals between six months and eighty years of age. The EPA incorporated additional data concerning the variability in CYP2E1 activity among humans, based on Lipscomb et al. (2003). The Lipscomb et al. (2003) study was based on in vitro analysis of liver samples from 75 human tissue donors (activity towards trichloroethylene and measurements of protein content) to estimate a distribution of activity in the population. These data support a wider distribution in CYP2E1 activity than had been used in the David et al. (2006) model, with approximately a sixfold range between the upper and lower bounds in Lipscomb et al. (2003) and a twofold range in David et al. (2006). Thus the EPA replaced the David et al. distribution parameters by using the same GM = 9.34, but GSD = 1.73. Further, since even the data available to Lipscomb et al. (2003) were limited, and the log-normal distribution is naturally bounded to be greater than zero, the EPA chose to use a non-truncated distribution for this parameter. (Since the distribution form for CYP2E1 was set by David et al. (2006) to be log-normal and the U.S. EPA chose to retain that form, even without specific bounds the distribution only includes values greater than zero.) Finally, the scaling of CYP2E1 for individuals under the age of 18 was adjusted based on the data of Johsrud et al. (2003); the EPA's analysis of these data indicate CYP2E1 activity in children is better predicted when assumed to scale with body weight (BW) raised to the 0.88 power, as compared to the more general power of 0.74, used by David et al. CYP2E1 activity for individuals over the age of 18 is still assumed to scale as $BW^{0.74}$.

Table 3-8. Parameter distributions used in human Monte Carlo analysis for dichloromethane by David et al. (2006)

Parameter		Distribution		Source
		Mean (arithmetic)	SD	
BW	Body weight (kg)	70.0	21.0	Humans[a]
QCC	Cardiac output (L/hr/kg$^{0.74}$)	16.5	1.49	Humans[a]
VPR	Ventilation:perfusion ratio	1.45	0.203	Humans[a]
QFC	Fat	0.05	0.0150	Humans[a]
QLC	Liver	0.26	0.0910	Humans[a]
QRC	Rapidly perfused tissues	0.50	0.10	Humans[a]
QSC	Slow perfused tissues	0.19	0.0285	Humans[a]
Tissue volumes (fraction BW)				
VFC	Fat	0.19	0.0570	Humans[a]
VLC	Liver	0.026	0.00130	Humans[a]
VLuC	Lung	0.0115	0.00161	Humans[a]
VRC	Rapidly perfused tissues	0.064	0.00640	Humans[a]
VSC	Slowly perfused tissues (muscle)	0.63	0.189	Humans[a]
Partition coefficients				
PB	Blood:air	9.7	0.970	Humans[b]
PF	Fat:blood	12.4	3.72	Rats[b]
PL	Liver:blood	1.46	0.292	Rats[b]
PLu	Lung:arterial blood	1.46	0.292	Rats[b]
PR	Rapidly perfused tissue:blood	1.46	0.292	Rats[b]
PS	Slowly perfused tissue (muscle:blood)	0.82	0.164	Rats[b]
Metabolism parameters				
$V_{max}c$	Maximum metabolism rate (mg/hr/kg$^{0.7}$)	9.42	1.23	Calibration[c]
K_m	Affinity (mg/L)	0.433	0.146	Calibration[c]
A1	Ratio of lung V_{Max} to liver V_{max}	0.000993	0.000396	Calibration[c]
A2	Ratio of lung KF to liver KF	0.0102	0.00739	Calibration[c]
FracR	Fractional CYP2E1 capacity in rapidly perfused tissue	0.0193	0.0152	Calibration[c]
First order metabolism rate (/hr/kg$^{0.3}$)				
	Homozygous (–/–)	0	0	Calibration[c]
k_fC	Heterozygous (+/–)	0.676	0.123	Calibration[c]
	Homozygous (+/+)	1.31	0.167	Calibration[c]

[a]US EPA, 2000d. Human PBTK model used for vinyl chloride.
[b]Andersen et al. (1987). Blood:air partition measured using human samples; other partition coefficients based on estimates from tissue measures in rats.
[c]Bayesian calibration based on five data sets (see text for description); posterior distributions presented in this table.

Source: David et al. (2006).

In addition, while the BW distribution in the David et al. (2006) PBTK model used ranges from 7 to 130 kg, thus covering 6-month-old children to obese adults, there are age-dependent changes and gender-dependent differences in ventilation rates and body fat that are not explicitly included. To more accurately reflect the distribution of physiological parameters in the entire population, EPA replaced the unstructured distributions of David et al. (2006) with distributions based on available information that specifically account for population variability in age, gender, and age- and gender-specific distributions or functions for BW, QCC, alveolar

ventilation, body fat (fraction), and liver fraction (see Appendix B for more details of the evaluation of each of these parameters).

The resulting set of parameter distribution characteristics, including those used as defined by David et al. (2006) are described in Table 3-9. Using this revised set of distributions, including the (revised) CYP and (published) GST activity distributions, and other distributions used as defined by David et al. (2006), the model as applied should reflect the full variability in the (U.S.) human population.

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Table 3-9. Parameter distributions for the human PBTK model for dichloromethane used by EPA

Parameter		Distribution: Shape	(Geometric) mean [a]	SD/GSD [a]	Lower bound	Upper bound	Section or source
BW	Body weight (kg)	Normal	*f (age, gender)*		*1st %tile*	*99th %tile*	B-4.3; NHANES IV
Flow rates							
QAlvC	Alveolar ventilation (L/hour/kg$^{0.75}$)	Normal	*f (age, gender)*	*f (age)*	*5th %tile*	*95th %tile*	B-4.4; mean: Clewell et al. (2004); SD: Arcus-Arth and Blaisdell (2007)
vprv	Variability in ventilation:perfusion ratio	Log-normal	1.00	0.203	0.69	1.42	VPR/VPR$_{mean}$ of David et al. (2006)
QCC	Cardiac output (L/hour/kg$^{0.75}$)	QCC$_{mean}$ = *f*(QAlvC)		QCC = QCC$_{mean}$/vprv			B-4.5; Clewell et al. (2004) (mean)
Fractional flow rates (fraction of QCC)							
QFC	Fat	Normal	0.05	0.0150	0.0050	0.0950	David et al. (2006); *after sampling from these distributions, normalize:* $Qi = \frac{QC \cdot QiC}{\sum QjC}$
QLC	Liver	Normal	0.26	0.0910	0.010	0.533	
QRC	Rapidly perfused tissues	Normal	0.50	0.10	0.20	0.80	
QSC	Slow perfused tissues	Normal	0.19	0.0285	0.105	0.276	
Tissue volumes (fraction BW)							
VFC	Fat	Normal	*f (age, gender)*	0.3·mean	0.1·mean	1.9·mean	Fat mean: B-4.6 (Clewell et al., 2004); liver mean: B-4.7 (Clewell et al., 2004); otherwise, David et al. (2006); *after sampling from these distributions, normalize:* $Vi = \frac{0.9215 \cdot BW \cdot ViC}{\sum VjC}$
VLC	Liver	Normal	*f (age*)	0.05·mean	0.85·mean	1.15·mean	
VLuC	Lung	Normal	0.0115	0.00161	0.00667	0.0163	
VRC	Rapidly perfused tissues	Normal	0.064	0.00640	0.0448	0.0832	
VSC	Slowly perfused tissues	Normal	0.63	0.189	0.431	0.829	
Partition coefficients							
PB	Blood:air	Log-normal	9.7	1.1	7.16	13.0	Geometric mean & GSD values listed here, converted from arithmetic mean and SD values of David et al. (2006)
PF	Fat:blood	Log-normal	11.9	1.34	4.92	28.7	
PL, PLu, & PR	Liver:blood, lung:arterial blood, and rapidly perfused tissue:blood	Log-normal	1.43	1.22	0.790	2.59	
PS	Slowly perfused tissue (muscle):blood	Log-normal	0.80	1.22	0.444	1.46	
Metabolism parameters (based on Monte Carte calibration from five human data sets)							
$V_{max}c$	Maximum metabolism rate (mg/hr/kg^{Xvmax})	Lognormal	9.34	1.73	(none)	(none)	B-3 mean: David et al. (2006); GSD & bounds: Lipscomb et al. (2003); Xvmax = 0.88 for age <18;. Xvmax = 0.70 for age ≥ 18.
K_m	Affinity (mg/L)	Log-normal	0.41	1.39	0.154	1.10	Geometric mean & GSD values listed here,

Table 3-9. Parameter distributions for the human PBTK model for dichloromethane used by EPA

Parameter		Distribution					Section or source
		Shape	(Geometric) mean [a]	SD/GSD [a]	Lower bound	Upper bound	
A1	Ratio of lung V_{Max} to liver V_{Max}	Log-normal	0.00092	1.47	0.000291	0.00292	converted from arithmetic mean and SD values of David et al. (2006)
A2	Ratio of lung KF to liver KF	Log-normal	0.0083	1.92	0.00116	0.0580	
FracR	Fractional MFO capacity in rapidly perfused tissue	Log-normal	0.0152	2.0	0.00190	0.122	
First order metabolism rate ([hour/$kg^{0.3}$]$^{-1}$)							
k_fC	Homozygous (–/–)	Normal	0	0	0	0	David et al. (2006)
	Heterozygous (+/–)	Normal	0.676	0.123	0.00	1.291	
	Homozygous (+/+)	Normal	1.31	0.167	0.00	2.145	

[a]Arithmetic mean and SD listed for normal distributions; geometric mean and geometric SD (GSD) listed for log-normal distributions.

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3.5.3. Evaluation of Rat PBTK Dichloromethane Models

Several deterministic PBTK rat models have been reported in the scientific literature (Sweeney et al., 2004; Andersen et al., 1991, 1987; Reitz , 1991; Reitz et al.,1988a, b; U.S. EPA 1988b, 1987a, b; Gargas et al., 1986). Unlike the mouse (Marino et al., 2006) and human (David et al., 2006), no hierarchical population model for dichloromethane in the rat exists in which parameter uncertainty is quantitatively integrated into model calibration. Rat data are not available that would allow for Bayesian calibration of individual metabolic parameters for the CYP or GST pathways. Thus, EPA assessed modified versions of deterministic rat PBTK models to select the most appropriate model for use in extrapolating internal dosimetry from rats to humans, for example in the determination of RfDs and RfCs based on effects seen in the rat. This work is described in detail in Appendix C and is based on evaluation of blood levels of dichloromethane and the percent saturation of hemoglobin as COHb (%COHb) and expired dichloromethane following intravenous injection (Angelo et al., 1986b), closed chamber gas uptake (Gargas et al., 1986), and dichloromethane and %COHb blood levels from a 4-hour inhalation exposure (Andersen et. al., 1991, 1987). Based on this work, the basic model structure of Andersen et al. (1991) was chosen, with the inclusion of lung dichloromethane metabolism via CYP (4% of liver metabolite production) and GST (14% of liver metabolite production) pathways (estimated from Reitz et al., 1989) (Figure 3-5) with metabolic parameters recalibrated against data of Andersen et al. (1991), based on prediction agreement of the various parameters with the available rat data sets. Table 3-10 presents the parameter distribution data for this model.

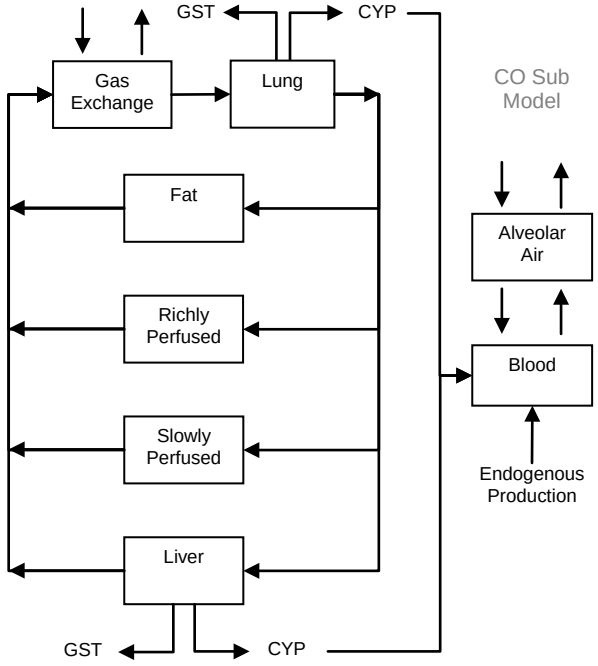

Figure 3-5. Schematic of rat PBTK model used in current assessment.

Table 3-10. Parameter values for the rat PBTK model for dichloromethane used by EPA

Parameter	Mean
Flow rates	
QCC (L/hour/kg$^{0.74}$)	15.9
VPR	0.94
Fractional flow rates (percent of QCC)	
Fat	9
Liver	20
Rapidly perfused tissues	56
Slowly perfused tissues	15
Tissue volumes (percent BW)	
Fat	7
Liver	4
Lung (scaled as BW$^{0.99}$)	1.15
Rapidly perfused tissues	5
Slowly perfused tissues	75
Partition coefficients	
Blood:air	19.4
Fat:blood	6.19
Liver:blood	0.732
Lung:arterial blood	0.46
Rapidly perfused tissue:blood	0.732
Slowly perfused tissue (muscle):blood	0.408
Metabolism parameters	
Maximum metabolism rate (mg/hour/kg$^{0.7}$)	3.93
Affinity (mg/L)	0.524
Ratio of lung V_{Max} to liver V_{Max}	0.04
Ratio of lung KF to liver KF	0.14
1st order metabolism rate (liver KF) ([hour/kg$^{0.3}$]$^{-1}$)	2.46
Oral absorption constant, k_a (1/hr)	1.80

3.5.4. Comparison of Mouse, Rat and Human PBTK Models

The comparison of various parameters across species (Table 3-11) primarily shows the modest inter-species differences that are known to occur in physiological parameters, also including the approximately 2-fold differences in partition coefficients which occur because of differences in rodent versus human blood lipid content. The 2.5-fold lower V_{max}c (CYP activity) in rats versus mice is also typical. The most striking difference is the variation in A1 and A2. Those values, however, reflect the in vitro differences originally quantified by Lorenz et al. (1984) and used in the dichloromethane PBPK modeling of Anderson et al. (1987). Thus these differences are based on independent measurements of tissue-specific metabolic capacity, and

while the specific values for mouse and human were refined through Bayesian analysis, the ultimate (posterior) values used are within a reasonable range of the in vitro measurements and so do not appear to be artifactual. (Since in vivo kinetics often indicate some differences from what would be predicted without adjustment from in vitro, it is not surprising that such differences occur here.) These differences do explain why lung-specific metrics in particular lead to lower internal dose and hence risk predictions in humans compared to whole-body metrics.

Table 3-11. Parameters in the mouse, rat, and human PBTK model for dichloromethane used by the EPA

Parameter	Mouse [a] Mean	Rat [b] Value	Human [c] Mean	Human [c] CV/*GSD* (Shape, bounds)	Sources
Fractional flow rates (fraction of cardiac output) [b]					David et al. (2006); *then normalized:* $Qi = \frac{QC \cdot QiC}{\sum QjC}$
QFC Fat	0.05	0.09	0.05	0.3 (N, 0.1-1.9)	
QLC Liver	0.24	0.20	0.26	0.35 (N, 0.0385-2.05)	
QRC Rapidly perfused tissues	0.52	0.56	0.50	0.2 (N, 0.4-1.6)	
QSC Slowly perfused tissues	0.19	0.15	0.19	0.15 (N, 0.553-1.453)	
Fractional tissue volumes (fraction of body weight) [b]					Fat mean: §2.2.3.6; Liver mean: §2.2.3.7; otherwise David et al. (2006); *then normalized:* $Vi = \frac{0.9215 \cdot BW \cdot ViC}{\sum VjC}$
VFC Fat	0.04	0.07	*f* (age, gender)	0.3 (N, 0.1-1.9)	
VLC Liver	0.04	0.04	*f* (age)	0.05 (N, 0.85-1.15)	
VLuC Lung	0.0115	0.0115	0.0115	0.14 (N, 0.58-1.42)	
VRC Rapidly perfused tissues	0.05	0.05	0.064	0.1 (N, 0.7-1.3)	
VSC Slowly perfused tissues	0.78	0.75	0.63	0.3 (N, 0.684-1.32)	
Partition coefficients [c]					Geometric mean (GM) & GSD/GM values converted from arithmetic mean & SDs of David et al. (2006)
PB Blood/air	23.0	19.4	9.7	*1.1* (LN, 0.738-1.34)	
PF Fat/blood	5.1	6.19	11.9	*1.34* (LN, 0.413-2.41)	
PL Liver/blood	1.6	0.73	1.43	*1.22* (LN, 0.552-1.81)	
PLu Lung/blood	0.46	0.46	1.43	"	
PR Rapidly perfused/blood	0.52	0.73	1.43	"	
PS Slowly perfused/blood	0.44	0.41	0.80	*1.22* (LN, 0.555-1.83)	
Flow rates					QCC: §2.2.3.5; vprv = VPR/VPR_{mean}: David et al. (2006); QAlvC: §2.2.3.4;
QCC Cardiac output (L/hr/$kg^{0.74}$)	24.2	14.99	QCC_{mean} = *f*(QAlvC)	QCC = QCC_{mean}/vprv	
VPR ventilation/perfusion ratio	1.45	0.94	(variable)	*(varies)* (LN, 0.69-1.42)	
QAlvC	QCC/VPR	QCC/VPR	*f (age, gender)*	*f (age)* (N, 5^{th}-95^{th} %)	
Metabolism parameters					V_{max}c: §2.2.2; others: David et al. (2006) (GM & GSD/GM values converted from arithmetic mean & SDs)
V_{max}c Maximum CYP metabolic rate (mg/hr/kg^{Xvmax})	9.27	3.93	9.34	*1.73* (LN, [unbounded])	
Xvmax CYP allometric scaling power	0.7	0.7	0.88 for age <18; 0.7 for age	*1.39* (LN, 0.376-2.68)	
K_m CYP affinity (mg/L)	0.574	0.524	0.41	-/-: NA	
k_fC First-order GST metabolic rate constant ($kg^{0.3}$/hr)	1.41	2.46	0 (-/-)[e]	+/-: 0.182 (N, 0-1.91)	
			0.676 (+/-)[e]	+/+: 0.128 (N, 0-1.64)	
			1.31 (+/+)[e]	*1.47* (LN, 0.316-3.17)	
A1 Ratio of lung V_{max}c to liver V_{max}c	0.207	0.04	0.00092	*1.92* (LN, 0.140-6.99)	
A2 Ratio of lung k_fC to liver k_fC	0.196	0.14	0.0083		

[a] Based on Marino et al. (2006) (source for all mouse parameters)

[b] Based on Andersen et al. (1991), with the addition of lung metabolism of dichloromethane via the CYP (4% of liver metabolite production) and GST (14% of liver

metabolite production) pathways. Physiological parameters and partition coefficients are from Andersen et al. (1991). The values for dichloromethane metabolism in the lung (as a fractional yield of liver metabolism for each pathway) were estimated from the in vitro ratios of enzyme activity (nmol/min/mg protein) in lung and liver cytosolic (GST) and microsomal (CYP) tissue fractions (Reitz et al., 1989). Metabolic parameters were re-optimized against the inhalation data of Andersen et al. (1991) using a heteroscedasticity parameter value of 2, which uses relative error for the model fitting algorithm. See Appendix C for further details.

[c] Based on David et al. (2006), with changes as noted. Additional sources include Clewell et al. (2004), Arcus-Arth and Blaisdell (2007), and Lipscomb et al. (2003). See identified sections for details. Distribution values (mean and a measure of dispersion) are provided with the CV (mean/SD) presented for normal (N) distributions and the GSD (italicized) presented for log-normal (LN) distributions. Distributions were truncated, bounds are (upper-lower bound)/mean.

[e] Values for the homozygous (–/–), heterozygous (+/–), and homozygous (+/+) GST-T1 genotypes, respectively.

HAZARD IDENTIFICATION

4.1. STUDIES IN HUMANS

4.1.1. Introduction—Case Reports, Epidemiologic, and Clinical Studies

There has been considerable interest in the influence of occupational exposure to dichloromethane in relation to a variety of conditions. The recognition that dichloromethane can be metabolized and bound to hemoglobin to form COHb, resulting in a reduction in the oxygen carrying capacity of the blood (Stewart et al., 1972b), prompted investigations into risk of ischemic heart disease and other cardiovascular effects. Reports of neurological effects from acute, high-exposure situations contributed to concern about neurological effects of chronic exposure to lower levels of dichloromethane. A general interest in potential cancer risk became more focused on lung and liver cancer because of the observation of these specific tumors in the NTP (1986) experiments in mice. Details of the studies pertaining to the experimental and epidemiologic studies of noncancer outcomes (e.g., cardiac, neurologic, hepatic, reproductive) are presented in section 4.1.2, and studies of cancer risk are presented in section 4.1.3.

4.1.2. Noncancer Studies

4.1.2.1. *Case Reports of Acute, High-dose Exposures*

Numerous case reports have been published that describe health effects resulting from acute exposure to dichloromethane. Most of the reports describe health effects resulting from inhalation of dichloromethane or dermal contact, but a few involve ingestion. The COHb levels in some of these cases were relatively low (7.5–13%), so the initial toxic effects of acute dichloromethane exposure appear to be due to its anesthetic properties as opposed to metabolic conversion of dichloromethane to CO.

Bakinson and Jones (1985) reported on a series of 33 cases of acute inhalation exposures to dichloromethane that occurred in the workplace over the period 1961–1980. Thirteen had lost consciousness, and one of the workers died. Nineteen cases reported general neurological effects, 13 reported gastrointestinal symptoms, 4 reported respiratory symptoms, and 1 reported hepatic symptoms. Of the 19 with general neurological symptoms, all reported headache, and dizziness was reported by 11 workers. Five workers reported one of the following symptoms: drunkenness, confusion, lack of coordination, or paresthesia.

Rioux and Myers (1988) summarized the health effects reported for 26 cases of dichloromethane poisoning published in the literature between 1936 and 1986. Three cases resulted from abuse-related exposures, 2 from chronic exposures, and 21 from acute exposures. The most common effects involved the central nervous system (CNS) (unconsciousness, drowsiness, headache, and behavioral symptoms), pulmonary edema and dyspnea, and dermatologic symptoms. Even severe symptoms could be reversed, but four deaths occurred.

More than 10 other case reports of fatalities or poisonings have been published since the summaries by Rioux and Myers (1988) and Bakinson and Jones (1985), and many of these incidents involve inadequately ventilated occupational settings (Jacubovich et al., 2005; Raphael et al., 2002; Fechner et al., 2001; Zarrabeitia et al., 2001; Goulle et al., 1999; Mahmud and Kales, 1999; Kim et al., 1996; Tay et al., 1995; Manno et al., 1992; Leikin et al., 1990; Shusterman et al., 1990). CNS depression and resulting narcosis, respiratory failure, and heart failure are common features of these reports. In a survey of workers in furniture stripping shops, 10 of the 21 workers stated that they sometimes experienced dizziness, nausea, or headache during furniture stripping operations (Hall and Rumack, 1990).

Chang et al. (1999) reported details of six patients who had ingested dichloromethane (four in a suicide attempt and two from accidental ingestion during a state of intoxication). The estimated amounts ingested were 350 mL or less. COHb levels, which were measured in only two of the cases, were 8.4 and 35% (with the latter being seen in a fatal case). As in exposures resulting from inhalation, the most common symptoms involved CNS depression, ranging from somnolence and weakness to deep coma. Tachypnea (n = 6) and corrosive gastrointestinal tract injury (n = 3) were also reported. Hepatic and renal failure and pancreatitis were found in the two most severe cases.

4.1.2.2. *Controlled Experiments Examining Acute Effects*

Several controlled experiments were conducted in the 1970s, examining neurophysiological effects and levels of COHb resulting from short-term (1–4 hours) exposures to dichloromethane at levels up to 1,000 ppm or longer-term exposures at levels up to 500 ppm. The 8-hour threshold limit value before 1975 was 500 ppm (National Institute of Occupational Safety and Health [NIOSH], 1986). These studies are described below. With the exception of Putz et al. (1979), there is no description in the published reports of the informed consent and other human subjects research ethics procedures undertaken in these studies, but there is no evidence that the conduct of the research was fundamentally unethical or significantly deficient relative to the ethical standards prevailing at the time the research was conducted.

In 1972, Stewart et al. (1972a, b) reported results from four experiments that were initiated because of the chance observation of an elevation in COHb saturation levels in an individual (one of the investigators) the morning after he had spent 2 hours working with varnish remover. Participants were medical students and faculty (including at least one of the coauthors). A total of 11 healthy nonsmoking volunteers were placed in an exposure chamber with mean concentrations of dichloromethane ranging from 213 to 986 ppm for 1 or 2 hours. These experiments indicated that dichloromethane exposure at these levels resulted in COHb saturation levels that exceeded and were more prolonged than those seen with threshold limit value exposures to CO. The exposures also resulted in symptoms of CNS depression indicated by visual evoked response changes and reports of light-headedness. Although return of COHb

levels to background levels could take >24 hours, all of the other symptoms were reversible within a few hours after exposure ceased.

Winneke (1974) measured auditory vigilance, visual flicker fusion frequency, and 14 psychomotor tasks in a total of 38 women exposed to dichloromethane levels of 300–800 ppm for 4 hours in an exposure chamber. A comparison group (nine females, nine males) exposed to 100 ppm CO for 5 hours was also included. Exposure to 800 ppm dichloromethane resulted in a statistically significant decrease in the performance of 10 of the 14 psychomotor tasks. In tests of auditory vigilance and visual flicker fusion, depressed response was seen at 300 ppm and was further depressed at 800 ppm. These effects were not seen with CO exposure.

Forster et al. (1974) exposed four healthy young men to dichloromethane levels ranging from 0 to 500 ppm for 7.5 hours/day for a total of 26 days over a 6-week period to investigate alterations in hemoglobin affinity for oxygen and altered pulmonary function. While no changes were observed in pulmonary function, hemoglobin affinity for oxygen was increased with no indication of adaptation to restore this affinity for oxygen to normal.

Putz et al. (1979) examined the behavioral effects seen after exposure to dichloromethane and to CO. Twelve healthy volunteers (six men and six women) each acted as his/her own control in separate 4-hour exposures to 70 ppm CO and 200 ppm dichloromethane. These levels were chosen so that the COHb level would reach 5% from both the CO and dichloromethane exposures. The experiments were conducted in a double-blind manner so that neither the investigators nor the participant knew the exposure condition under study at any particular time. Informed consent was obtained, and the study was reviewed by the NIOSH Human Subject Review Board. The performance tests were dual tasks (an eye-hand coordination task in conjunction with a tracking task), with five measures of performance assessed at six time points over the 4-hour test period and an auditory vigilance task. Two levels of difficulty were assessed for each task to allow assessment of whether the exposure effect was similar in low and high difficulty tasks. The tests of eye-hand coordination, tracking tasks, and auditory vigilance revealed significant impairment with both exposures under the more difficult task conditions. Effects were similar or stronger in magnitude for dichloromethane compared with CO.

4.1.2.3. *Observational Studies Focusing on Clinical Chemistries, Clinical Examinations, and Symptoms*

Studies in currently exposed workers

Ott et al. (1983a, c, d) evaluated several parameters of hepatic, hematopoietic, and cardiac function in workers exposure to dichloromethane in a triacetate fiber production plant in Rock Hill, South Carolina. Two hundred sixty-six Rock Hill workers and a comparison group of 251 workers in an acetate fiber production plant in Narrows, Virginia, were included in the examination of urinary and blood measures. These groups included men and women, blacks and whites, and smokers and nonsmokers. The median 8-hour TWA exposure for dichloromethane

ranged from 60 to 475 ppm in Rock Hill. Acetone at levels up to over 1,000 ppm was present in both plants, but dichloromethane and acetone exposures were inversely related.

There were differences in blood collection procedures between the two plants and in the age, sex, race, and smoking history distribution of the study groups. The demographic and smoking differences were accounted for in the analysis by stratification. Statistically significant differences were seen between the workers in the two plants for COHb, serum alanine aminotransferase (ALT), total bilirubin, and mean corpuscular hemoglobin concentration (MCHC) (although the direction and magnitude of these differences were not reported and the authors stated that the difference in serum ALT could be due to the differences in blood collection procedures, which involved a sitting versus recumbent position of the subjects at the exposed and nonexposed plants, respectively) (Ott et al., 1983c). Within the Rock Hill plant, analyses were also conducted to examine associations between dichloromethane exposure and the clinical parameters within specific race-sex groups by using multiple regression to control for smoking status, age, and time of venipuncture. Positive associations were seen with COHb in all race-sex groups (increases of 0.7–2.1% per 100 ppm increase in dichloromethane) and with total bilirubin (increases of 0.05–0.08 mg/dL per 100 ppm increase in dichloromethane) in all groups except nonwhite men (which was a much smaller group, n = 20, than the other groups). Red cell count, hematocrit, hemoglobin, and aspartate aminotransferase (AST) were also positively associated with dichloromethane exposure in white females. The increase in total bilirubin level was not supported by parallel changes in other measures of liver function or red blood cell turnover, suggesting that this measure was not reflecting liver damage or hemolysis.

The increased red cell count, hemoglobin, and hematocrit in women exposed to high levels of dichloromethane (up to 475 ppm, 8-hour TWA) may indicate a compensatory hematopoietic effect. The fact that these changes were not significant among men may be due to higher baseline hemoglobin, which was observed when comparisons were made between nonsmoking men and women. No such difference in the baseline values was observed among the smoking men and women, suggesting that the compensatory advantage may be lost among smokers.

Ott et al. (1983e) present results from a further investigation of changes in COHb, alveolar CO, and oxygen half-saturation pressure in relation to dichloromethane exposure. Blood samples were collected before and after shift from 136 Rock Hill and 132 Narrows workers. For the Rock Hill workers, personal monitoring for dichloromethane exposure was done during the shift. The TWA for dichloromethane ranged from 0–900 ppm, with a bimodal distribution (peaks around 150 and 500 ppm) resulting from the layout of the plant. The blood samples were used to determine blood COHb and alveolar CO levels, and the partial oxygen pressure (P_{50}) (that is, the pressure required to keep 50% of the blood oxygen-carrying capacity saturated with oxygen at pH 7.4 and 37°C). Separate analyses were conducted for smokers and nonsmokers to account for the smoking-related effects on COHb. Linear relationships were seen

between dichloromethane exposure and the before-shift COHb and alveolar CO levels, reflecting residual CO metabolism from the previous day's exposure. There were significant quadratic relationships between dichloromethane exposure and the postshift COHb and alveolar CO levels, indicating a partial saturation of the enzyme system metabolizing dichloromethane. The P_{50} group means were lower among the exposed compared with the referents, among smokers compared with nonsmokers, and among men compared with women. Given the relationship between COHb and P_{50}, an expected decrease in P_{50} during the shift was observed among the exposed.

Continuous 24-hour cardiac monitoring was also evaluated in a smaller sample of 24 dichloromethane-exposed workers from the triacetate fiber production plant in Rock Hill, South Carolina, and 26 workers from the comparison plant in Narrows, Virginia. This study (Ott et al., 1983d) was limited to white men ages 35 or more years. Special efforts were made to recruit men with a history of heart disease, because this group was postulated to be most likely to demonstrate positive findings. The estimated TWA dichloromethane exposure ranged from 60 to 475 ppm in the exposed group. The evaluation examined ventricular and supraventricular ectopic activity and S-T segment depression in the exposed and nonexposed groups. Comparisons were also made between cardiac performance during work hours and nonwork hours to discern possible short-term effects of recent exposure. Comparing the findings for the 24 exposed and 26 referent volunteers indicated no difference in ventricular or supraventricular ectopic activity or S-T-segment depression. There was no difference comparing work and nonwork hours among exposed.

Soden et al. (1996) studied all active male workers exposed to dichloromethane at a Hoechst Celanese triacetate film production plant in Belgium. The production process was the same as the process at the Hoechst Celanese Rock Hill plant, except the Belgium plant was newer with better engineering controls to significantly reduce overall levels of the dichloromethane, acetone, and methanol used in the process. The objectives of the study were to determine the impact of varying levels of dichloromethane exposure on COHb levels, whether successive days of dichloromethane exposure affected the COHb levels, and what impact smoking had on COHb levels in conjunction with dichloromethane exposure. Workers were monitored semiannually for COHb at the end of the work shift and were personally monitored for exposure to the three solvents. Smoking status was defined based on a health assessment questionnaire, with smokers smoking at least one cigarette per day. Among nonsmokers, a dose response was found among COHb levels and average dichloromethane exposure levels in the range of 7–90 ppm. The maximum COHb was 4.00% at an average exposure of 90 ppm (correlation coefficient = 0.58, $p < 0.05$). Smokers' COHb levels were elevated when compared with those of nonsmokers with similar dichloromethane air levels, but the dose-response correlation between dichloromethane air levels and COHb levels was weaker and not statistically significant (correlation coefficient 0.20). The maximum COHb level for smokers was 6.35% at

an average dichloromethane air level of 99 ppm. The authors concluded that dichloromethane exposures up to the levels observed did not produce COHb levels that are likely to cause cardiac symptoms.

Cherry et al. (1983, 1981) reported the results of health evaluations of two studies of triacetate film production workers. Cherry et al. (1981) recruited 46 of the 76 male workers at a triacetate film factory, where workers were exposed to dichloromethane and methanol in a ratio of 9:1 at air levels of dichloromethane ranging from 75 to 100 ppm. A small comparison group (n = 12) of workers at this factory who worked a similar shift pattern (rapidly rotating shifts) but who were not exposed to dichloromethane was also included. The men were asked whether they had ever experienced cardiac symptoms (pain in the arms, chest pain sitting or lying, or chest pain when walking or hurrying) and were asked about the presence, in the past 12 months, of neurological disorders (frequent headaches, dizziness, loss of balance, difficulty remembering things, numbness and tingling in the hands or feet), affective symptoms (irritability, depression, tiredness), and stomachache (as an indicator of symptom overreporting). No difference in response was found in history of stomachache (reported by 15% of exposed workers compared with 17% nonexposed workers). Six of the exposed and none of the unexposed men responded positively to the cardiac symptoms. The exposed group reported an excess of neurological symptoms; the number (and proportion) reporting zero, one, two, and three or more symptoms were 26 (0.56), 8 (0.17), 9 (0.20), and 3 (0.07), respectively, in exposed workers compared with 11 (0.92), 1 (0.12), 0 (0.00), and 0 (0.00), respectively, in controls ($p < 0.02$ for chi-square test of linear trend). With respect to affective symptoms, the number (and proportion) reporting zero, one, two, and three symptoms were 28 (0.61), 6 (0.13), 7 (0.15), and 5 (0.11), respectively, among the exposed workers, and 9 (0.75), 2 (0.17), 1 (0.08), and 0 (0.0), respectively, among the unexposed workers. The authors concluded that there was no difference between exposed and nonexposed in reporting of affective symptoms based on a chi-square test of linear trend. There was no discussion of the statistical power of this test or of tests of the proportion reporting a specified number of symptoms (which may be a more appropriate test given the sample size), but it is clear that the statistical power of this test was very low. For example, taking the simple case of the comparison of the proportion reporting two or more symptoms and using the approximate estimates from this study (25 and 10% in the exposed and unexposed, respectively), approximately 75 exposed and 300 unexposed workers would be needed for a power of 0.80 (i.e., an 80% chance of rejecting the null hypothesis when the null hypothesis was false); the actual power with the sample size of 46 and 12 is less that 0.10.

Based on these results, a follow-up study was conducted, which included a larger referent group. This study included the symptom list described in the previous paragraph, a standardized clinical exam (including an electrocardiograph), and neurological and psychological tests of nerve conduction, motor speed and accuracy, intelligence, reading, and memory (Cherry et al., 1981). Twenty-nine of the original 46 exposed workers participated in the follow-up. The men

who did not participate in the follow-up were similar in age and symptoms to the men who did. The new referent group was recruited from another plant with the same owner and a very similar process but without dichloromethane exposure. One control, age-matched within 3 years, was selected for each exposed worker. No differences between the groups were found in the clinical exam, electrocardiogram, or nerve conduction tests. A statistically significant ($p < 0.05$) deficit among the exposed workers was found for coarse motor speed. On two tests of overall intelligence, the exposed group did significantly better than the referent, but, on a reading ability test designed to assess premorbid educational level, scores for the exposed group were slightly lower than for the referent group. (Only one of these three differences, the trail making intelligence test, was statistically significant.) With respect to the report of neurological symptoms in the past year, the number (and proportion) reporting zero, one, two, and three symptoms were 17 (0.59), 4 (0.14), 6 (0.21), and 2 (0.07), respectively, among the exposed workers, and 21 (0.72), 6 (0.21), 0 (0.0), and 2 (0.07), respectively, among the unexposed workers, with a test of linear trend that was not statistically significant. The authors interpret the results as indicating that the differences in neurological symptoms seen in the initial study were due to chance and that, taken as a whole, the exposed workers had no detrimental effect attributable to dichloromethane exposure. Again, the limitations of the statistical power of the analysis and alternative interpretations that might have resulted from approaches taken to improve the power were not discussed. These approaches include combining the unexposed groups from the two analyses, using the full sample of the exposed group instead of the subset of 29 who completed the clinical exam, or using a different test (i.e., of a proportion rather than a linear trend),

Cherry et al. (1983) compared dichloromethane-exposed workers at an acetate film factory to nonexposed workers (from the same plant but from areas without solvent contact or from another film production factory in which solvents were not used). The 56 exposed and 36 unexposed workers were matched to within 3 years of age. Both factories were on rapid rotating shifts. Exposure to dichloromethane ranged from 28–173 ppm, using individual air sampling pumps. Blood samples were taken to monitor dichloromethane levels at the beginning and end of the shift. Study participants were asked to rate sleepiness, physical and mental tiredness, and general health on visual analog scales with the extreme responses at either end. Participants were also given a digit symbol substitution test and a test of simple reaction time. No differences were seen between exposed and unexposed groups at the beginning of the shift on the four visual analogue scales, but the exposed deteriorated more on each of the scales than did the controls. This difference in deterioration was statistically significant ($p \leq 0.05$) during the morning shift but was not statistically significant during the afternoon or night shifts. A significant correlation was shown between change in mood over the course of the shift and level of dichloromethane in the blood. No difference was seen between the exposed and referents on the tests of reaction time or digit substitution. However, among the exposed, deterioration in the

digit substitution tests at the end of the shift was significantly related to blood dichloromethane levels.

Anundi et al. (1993) studied 12 men who worked in a graffiti-removing company. Each worker filled out a questionnaire about previous occupational and non occupational exposure to solvents and use of protective equipment. Half-day breathing zone samples were taken for each of the 12 workers, and 15-minute samples were also taken for 10 workers. On the day the air sampling was done, a structured interview pertaining to recent diseases or symptoms related to allergies, asthma, diseases of the skin, respiratory organs, gastrointestinal tract, urinary organs, neurological trauma and disease, and neuropsychiatric symptoms was conducted by a physician, and blood and urine samples were collected. The results were compared with those of 233 men from the area population. The 12 men (mean age 23 years) had worked between 3 months and 4.5 years cleaning graffiti from underground stations. No respiratory protection was used, and the leather gloves were frequently soaked with solvent. While mixed solvent was used to do the cleaning, dichloromethane was the predominant component, as confirmed by the air samples. The geometric mean of the TWA calculated from the half-day samples was 127 mg/m^3 (range 18–1,188 mg/m^3), with half of the samples exceeding the Swedish permissible exposure limit of 120 mg/m^3. The geometric mean of the 15-minute samples was 400 mg/m^3 (range 6–5,315 mg/m^3), with most samples exceeding the Swedish short-time exposure limit of 300 mg/m^3. Two workers had clinical laboratory data outside the normal range (urinary α_1- or β_2-microglobulin, serum ALT, γ-glutamyl transpeptidase), which could indicate possible kidney and liver damage. The authors stated that in both cases factors other than the solvent exposure (i.e., urinary tract medical condition preceding employment, history of renal stones) could have influenced these laboratory results. The prevalence of irritation of the eyes and upper respiratory tract (blocked nose and nasal catarrh) was much higher in the graffiti-cleaning workers compared with the referent group (e.g., >70% of the workers compared with 18% of the comparison group reported a blocked nose; ~50% of workers and 15% of the comparison group reported eye irritation), but there were no or much smaller differences in abnormal tiredness, headache, nausea, or irritative cough. No acute effects on the CNS were noted.

Studies in retired workers

Lash et al. (1991) examined the hypothesis that long-term exposure to dichloromethane produces lasting CNS effects as measured by long-term impairment on memory and attention centers. Retired aircraft maintenance workers employed in at least 1 of 14 targeted jobs with dichloromethane exposure for 6 or more years between 1970 and 1984 were compared to a like group of workers without dichloromethane exposure. The unexposed workers were also retired aircraft mechanics at the same base and held one of 10 jobs in the jet shop where little solvent was used. The exposed group made up of painters and mechanics in the overhaul department was chosen to maximize the exposure contrast yet minimize differences in potential confounders

between exposed and nonexposed. Exposures were typically within state and federal guidelines for dichloromethane exposure. From 1974 to 1986, when 155 measurements for dichloromethane exposure were made, mean breathing zone TWAs ranged from 82 to 236 ppm and averaged 225 ppm for painters and 100 ppm for mechanics.

Data collection occurred in three phases, with an initial questionnaire to all retired members of the airline mechanics union to identify eligible workers, followed by a telephone survey to collect medical, demographic, and general employment criteria. Subjects who qualified were then recruited to participate in the medical evaluation. Sixty percent of the 1,758 retirees responded to the questionnaire and 259 of these retirees met the eligibility criteria. Ninety-one men qualified for the medical evaluation based on the telephone survey; 25 retirees exposed to solvents, and 21 unexposed retirees participated in the evaluation. All were men between the ages of 55 and 75, without a history of alcoholism or any neurological disorder. The 25 exposed participants worked an average of 11.6 years in dichloromethane-exposed jobs during the target period and 23.8 years in the industry.

The medical evaluation included a questionnaire about the occurrence of 33 different symptoms in the past year, physiological measurement of odor and color vision senses, auditory response potential, hand grip strength, and measures of reaction time (simple, choice, and complex), short-term visual memory and visual retention, attention, and spatial ability. The only large differences (i.e., effect size, or mean difference between groups divided by the SD of the outcome measure, of 0.4 or greater) between the two groups were a higher score on verbal memory tasks (effect size approximately 0.45, $p = 0.11$) and lower score on attention tasks (effect size approximately –0.55, $p = 0.08$) and complex reaction time (effect size approximately –0.40, $p = 0.18$) in the exposed compared with the control group. (Although not noted by the authors, the power to detect a statistically significant difference between the groups, given this sample size, was low [i.e., approximately 0.30 for an effect size of 0.40, using a two-tailed alpha of 0.05]) (Cohen, 1987). The authors investigated the possibility of response bias, given the low initial response to the mailed questionnaire recruiting retirees and the small number of workers from the entire pool of eligible participants who actually participated in the medical evaluation. Attempts were made to contact 30% of the questionnaire nonrespondents, with 46% contacted and 31% completing the telephone interview. The only difference found between those who responded to the mailed questionnaire and those who did not was a higher percentage of diagnosed heart disease among the nonrespondents who were 2.5 years older and had been retired 1.7 more years than the respondents. Those who were eligible but did not participate in the medical evaluation were similar to the exam participants on all characteristics included in the interview. The only difference was a higher prevalence of gout among the unexposed who did not participate compared to the unexposed who did participate.

4.1.2.4. *Observational Studies Using Workplace Medical Program Data*

Kolodner et al. (1990) investigated the effect of occupational exposure to dichloromethane on six health outcomes identified in the literature or based on biological plausibility. Participants in the study were male workers at least 19 years old at two General Electric plastic polymer plants where dichloromethane was one of the chemicals used. Four dichloromethane exposure categories were established based on full-shift personal air monitoring data (8-hour TWA) collected in 1979–1985, job titles, and industrial hygienists' knowledge of plant operations. The mean 8-hour TWA and number of workers in each of the four exposure groups were 49.0 ppm for the 19 workers in the highest, 10.9 ppm for the 49 workers in the intermediate, 3.3 ppm for the 56 workers in the low, and <1.0 ppm for the 772 workers in the minimal/no exposure group.

Data from 1984 annual medical exams and 1985 absence data from payroll records were evaluated for possible health effects resulting from occupational exposure to dichloromethane. A high percentage of workers participated in the annual medical exams, with only 5 of the 896 eligible for inclusion in the study refusing the exam completely in 1984. Six hypotheses were specifically tested regarding dichloromethane exposure in relation to different health outcomes: absence due to illness, hepatotoxicity (manifested by nausea, weakness and fatigue, palpable liver, abdominal tenderness, jaundice, hepatomegaly, abnormal serum γ-glutamyl transferase, ALT, AST, or bilirubin), diabetes mellitus (manifested by weight loss, weakness and fatigue, polydypsia, polyuria, impaired vision, excessive weight loss, elevated fasting blood sugar, and abnormal urinary glucose or urinary acetone), CNS toxicity (manifested by headache, lightheadedness, dizziness and vertigo, ataxia, weakness and fatigue, and abnormalities detected in the central motor, central sensory, cranial nerve, gait, neurocoordination, or Bibinski reflex examinations), cardiovascular abnormalities (manifested by fatigue, dyspnea, chest pain with exertion, palpitations, or abnormalities detected in the point maximum impulse exam, blood pressure measurements, or electrocardiogram), and neoplastic breast changes (154 women were included in this portion of the study—manifested by painful breast, breast swelling, lump, nipple discharge, or abnormalities detected in the breast examination).

Workers were placed in exposure categories based on their current jobs. In addition, exposure to high noise levels occurred in both plants, and workers in each plant had exposure to another chemical, either phenol or phosgene. The authors noted that workers tended to move from entry-level jobs with high dichloromethane exposure to supervisory jobs with lower dichloromethane exposure, based on the seniority system in place at both plants. Thus, current exposure levels reported did not necessarily reflect cumulative exposure. Because of the way the seniority system moved workers through jobs and the fact that workers were assigned to dichloromethane exposure categories based on their current job, age was inversely related to exposure and was controlled in the analysis of some of the continuous variables using analysis of covariance. Age adjustment was not employed in the analysis of dichotomous variables. The

mean age was 35.3, 39.7, 37.1 and 29.5 years in the minimal/no, low, medium[3], and high exposure groups, respectively. The small number of workers in the exposed groups limited the ability to evaluate the effects of dichloromethane exposure on health outcomes related to age, since age had to be adjusted in these analyses. The racial distribution did not differ among the exposure groups.

The authors indicated that all the hypotheses were accepted with the exception of CNS symptoms. However, it should be noted that the small size and younger age distribution in the high exposure group and the lack of adjustment for age in most of the analyses make it difficult to interpret the statistical testing that was performed. Data pertaining to neurological, hepatic, and cardiac function are shown in Table 4-1. Among the six neurological symptoms evaluated, a statistically significant positive exposure-effect relationship between dizziness/vertigo and dichloromethane exposure was identified. This trend was driven most strongly by the low frequency of this reported symptom in the minimal/no exposure group (1.2%), but there was no linear trend across the higher levels of exposure (7.5, 2.1, and 5.3% in the low, medium, and high exposure groups, respectively).

[3] The "medium" exposure group is also referred to as the "intermediate" exposure group in Kolodner et al. (1990).

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Table 4-1. Percentage of male General Electric plastic polymer workers reporting neurologic symptoms or displaying abnormal values in measures of neurological function, hepatic function, and cardiac function

	Exposure Group[a]			
	Minimal/no (n = 772)	Low (n = 56)	Medium (n = 49)	High (n = 19)
Neurological				
Headache	8.7	7.5	10.4	5.3
Lightheadedness	2.9	3.8	4.2	5.3
Dizziness/vertigo	1.2	7.5	2.1	5.3
Ataxia	0.0	1.9	0.0	0.0
Babinsky	0.0	0.0	0.0	0.0
Gait	0.0	0.0	0.0	0.0
Faintness/syncope[b]	0.1	0.0	2.1	0.0
Seizures[b]	0.4	0.0	2.1	0.0
Paresis/paralysis[b]	0.7	0.0	0.0	0.0
Parasthesis[b]	4.0	7.5	14.6	0.0
Head trauma/concussion[b]	0.8	1.9	0.0	0.0
Peripheral motor exam[b,c]	0.5	0.0	0.0	0.0
Peripheral sensory exam[b,c]	1.1	2.4	5.1	0.0
Rhomberg exam[b,c]	0.0	0.0	2.6	0.0
Hepatic				
Serum gamma glutamyl transferase	8.0	16.1	12.2	5.3
Serum total bilirubin	3.0	1.8	2.0	10.0
Serum AST	1.8	3.6	4.1	0.0
Serum ALT	9.1	10.7	8.2	5.3
Cardiac[d]				
Palpitations: percent abnormal	1.2	9.1	2.1	0.0
Electrocardiogram				
borderline/abnormal	18.5	16.7	19.1	8.3
bradycardia/tachycardia abnormalities[b]	20.2	16.7	25.5	0.0
general rhythm abnormalities	12.0	11.1	17.0	8.3
atrial, atrioventricular, or sinus abnormalities	0.8	0.0	0.0	0.0
bundle blocks or ventricular abnormalities	3.9	5.6	10.6	8.3
axis deviations	2.6	1.9	2.1	8.3
wave abnormalities	4.0	3.7	10.6	0.0
hypertrophy	3.8	3.7	6.4	0.0
evidence of infarction	2.3	5.6	2.1	0.0

[a]Mean 8-hour TWA exposure was <1.0, 3.3, 0.9, and 49.0 ppm in the minimal/no, low, medium, and high groups, respectively; mean age 35.3, 39.7, 37.1, 29.5 years in the minimal/no, low, medium, and high groups, respectively.
[b]The authors considered these to be screening variables rather than hypothesis-testing variables.
[c]n = 629, 42, 39, and 14 in the minimal/no, low, medium, and high groups, respectively
[d]For all cardiac outcomes except bradycardia/tachycardia, n = 728, 54, 47, and 12 in the minimal/no, low, medium, and high groups, respectively. For bradycardia/tachycardia, n = 727 in the minimal/no group.

Source: Kolodner et al. (1990).

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2384 Soden (1993) compared health-monitoring data from dichloromethane-exposed workers
2385 in the Rock Hill triacetate fiber production plant to workers from another plant making polyester
2386 fibers owned by the same company in the same geographic area. Exposed and control workers
2387 were chosen from among workers who had worked at least 10 years in their respective areas and
2388 who participated in the company's health-monitoring program between 1984 and 1986 and were

still employed on December 31, 1986. Controls were matched by race, age, and gender to each Rock Hill worker for a sample size of 150 and 260 in the exposed and control groups, respectively. (The aim of the study had been 1:2 matching.) The 8-hour TWAs among the Rock Hill workers were those reported by Lanes et al. (1990), namely 475 ppm for dichloromethane, 900 ppm for acetone, and 100 ppm for methanol. None of these exposures occurred at the polyester plant. There was a 90% participation rate in the health-monitoring program. Six questions in the health history portion of the health-monitoring program concerned cardiac and neurological symptoms (chest discomfort with exercise; racing, skipping, or irregular heartbeat; recurring severe headaches; numbness/tingling in hands or feet; loss of memory; dizziness). Part of this program included blood samples used for standard clinical hepatic and hematologic parameters: serum ALT, AST, total bilirubin, and hematocrit. The clinical measures were available for 90 (60%) of the exposed and 120 (46%) of the control group because some participants declined this part of the health-monitoring program because similar tests had been part of recent personal medical care.

There was little difference in the frequency of reported symptoms between exposed workers and controls: chest discomfort reported by 2.0% of exposed and 4.0% of the controls, irregular heartbeat reported by 5.5% of exposed and 6.0% of the controls, recurring severe headaches reported by 3.5% of exposed and 5.5% of the controls, numbness/tingling in hands and feet reported by 6.4% of exposed and 8.1% of the controls, loss of memory reported by 1.3% of exposed and 0.4% of controls, and dizziness reported by 2.7% of exposed and 4.8% of controls (Soden, 1993). The levels of the blood values were similar in the exposed and control groups, except for a 3.1 IU/L decrease in serum AST activity ($p = 0.06$). The authors concluded that this difference was not clinically significant but did not discuss the potential bias introduced by the selective participation in this part of the study.

4.1.2.5. *Studies of Ischemic Heart Disease Mortality Risk*

Several studies have examined the relation between dichloromethane exposure and risk of cardiovascular-related mortality. The methodological details of these studies are described in section 4.1.3.2.). No evidence of increased risk of ischemic heart disease mortality was seen in two triacetate film production cohort studies (Hearne and Pifer, 1999; Tomenson et al., 1997) or in two triacetate fiber production cohort studies (Gibbs et al., 1996; Lanes et al., 1993). Information on this outcome was not included in the dichloromethane analysis of civilian Air Force base workers (Blair et al., 1998). The standardized mortality ratios (SMRs) for ischemic heart disease mortality were <1.0 in all of the cohorts and dose groups examined (Table 4-2). The "healthy worker effect" may have contributed to these observations. There are no case-control studies of ischemic heart disease and dichloromethane exposure.

Table 4-2. Ischemic heart disease mortality risk in four cohorts of dichloromethane-exposed workers

		Obs[a]	Exp[b]	SMR	95% CI[c]
Triacetate film production					
Hearne and Pifer (1999)	Cohort 1 (men)	117	136.7	0.86	0.71–1.03
	Cohort 2 (men)	122	143.3	0.85	0.71–1.02
Tomenson et al. (1997)	Men	114	123.9	0.92	0.76–1.10
Triacetate fiber production					
Lanes et al. (1993)	Men and women	43	47.8	0.90	65–121
Gibbs et al. (1996)	Men				
	50–100 ppm	96	100.1	0.96	0.78–1.2
	350–700 ppm	98	106.8	0.92	0.75–1.1
	Women				
	50–100 ppm	32	45.8	0.70	0.48–0.99
	350–700 ppm	0	3.4	–	0.0–1.1

[a]Obs = number of observed deaths.
[b]Exp = number of expected deaths.
[c]CI = confidence interval.

4.1.2.6. *Studies of Suicide Risk*

Suicide risk is not an outcome that was a primary hypothesis or motivation of the cohort studies but may be relevant given the potential neuropsychological effects of dichloromethane, as evidenced from studies of acute and chronic exposure scenarios described previously. In a triacetate film production cohort in Rochester, New York, Hearne and Pifer (1999) reported 14 observed deaths from suicide compared with 7.8 expected, for an SMR of 1.8 (95% confidence interval [CI] 0.98–3.0) (Table 4-3). This cohort ("Cohort 1") consisted of 1,311 men who were first employed between 1946 and 1970 and were followed through 1994. Similar results were seen in a different, but somewhat overlapping, cohort in this study ("Cohort 2") of 1,013 men employed between 1964 and 1970 and followed through 1994 (see section 4.1.3.3.1). There was also evidence of increasing suicide risk with dichloromethane exposure, particularly in the highest exposure group, in the study of triacetate fiber production workers in Maryland (Gibbs, 1992). The triacetate fiber production cohort study in Rock Hill, South Carolina, has published what appears to be erroneous information about suicide risk. In the 1993 paper (Lanes et al., 1993), 4 observed and 5.21 expected cases were reported (SMR 0.77), but the SMR that was reported with these data was 1.19 (95% CI 0.39, 2.8). This ratio would correspond to 6 observed and around 5.2 expected cases. Information on suicide was not included in the other film and fiber cohort studies (Tomenson et al., 1997) or in the analysis of civilian Air Force base workers (Blair et al., 1998). There are no case-control studies of suicide risk and dichloromethane exposure.

Table 4-3. Suicide risk in two cohorts of dichloromethane-exposed workers

		Obs[a]	Exp[b]	SMR	95% CI
Triacetate film production					
Hearne and Pifer (1999)	Cohort 1	14	7.8	1.8	0.98–3.0
	Cohort 2	9	5.1	1.8	0.81–3.4
Triacetate fiber production[c]					
Gibbs (1992)	50–100 ppm	8	6.4	1.3	0.54–2.5
	350–700 ppm	8	4.4	1.8	0.78–3.6

[a]Obs = number of observed deaths.
[b]Exp = number of expected deaths.
[c]One additional study provided data on suicide risk, but some kind of error seems to be present: 4 observed and 5.21 expected cases were reported in Lanes et al. (1993), which would be an SMR of 0.77, but the SMR that was reported with these data was 1.19 (95% CI 0.39, 2.8). This ratio would correspond to 6 observed and around 5.2 expected cases.

4.1.2.7 *Studies of Infectious Disease Risk*

There is limited information pertaining to infectious disease risk in relation to dichloromethane exposure. Only one of the cohort studies (Hearne and Pifer, 1999) reported data for the broad category of infectious and parasitic disease mortality. In Cohort 1 of this analysis, there were no observed deaths in this category (5.6 expected), and in Cohort 2 there were 3 observed and 4.7 expected, for an SMR of 0.64. The detailed report by Gibbs (1992) of the cellulose triacetate fiber production cohorts in Maryland (Gibbs et al., 1996) also contained information on the facility in South Carolina that was the site of the report by Lanes et al. (1993, 1990). Slightly elevated risks of mortality due to influenza and pneumonia were seen among the male workers in the high exposure group in Maryland (7 observed, 5.62 expected, SMR 1.25) and in South Carolina (3 observed, 1.33 expected, SMR 2.26). Among females, there were few observed or expected cases (in Maryland, 1 observed, 0.23 expected, SMR 4.36; in South Carolina, 0 observed and 0.74 expected).

4.1.2.8. *Studies of Reproductive Outcomes*

Pregnancy outcomes in women exposed to dichloromethane have been investigated in two studies. Taskinen et al. (1986) studied spontaneous abortions among women employed in eight pharmaceutical factories between 1973 and 1980. Data on pregnancy outcomes were collected from a national hospital and clinic discharge registry in Finland from 1973 to 1981 by matching the worker rosters to the registry. Exposure to dichloromethane was one of eight solvents or classes of solvents included in the study. The study consisted of two parts. The first investigated the rate of spontaneous abortions (number of spontaneous abortions divided by the sum of spontaneous abortions and births) during, before, or after employment in the pharmaceutical industry. One hundred and forty-two spontaneous abortions and 1,179 births were identified among the female workers at the eight plants. Employment hire and termination

dates were obtained from plant records. The spontaneous abortion rate was 10.9% during employment compared with 10.6% before and after employment. These results compared to a rate of 8.5% in the general population in the geographic area where the factories were located. The rate of spontaneous abortions among workers declined over the period of the study, with a 3-year moving average of 15% at the beginning declining to 9.5% at the end of the study. Over the same period, the industrial hygiene allegedly improved in the plants. Ten congenital malformations of different types were identified among the women (five among those who were employed in the pharmaceutical industry during the pregnancy and five among those whose pregnancies occurred before or after this employment).

The second part of the study by Taskinen et al. (1986) was a case-control study of the risk of spontaneous abortions in relation to workplace exposures during pregnancy. The source population consisted of women who were employed in one of the eight Finnish pharmaceutical factories during at least 1 week of the first trimester of pregnancy during the study period. Cases (n = 44) were selected from this population based on hospital or clinic records indicating a spontaneous abortion, and 130 controls (women who had given birth) were age-matched (3:1 matching, age within 2.5 years) to each case. Occupational exposure data were obtained by questionnaires completed by the plant physician or the nursing staff, blinded to the case status of the study member, in consultation with labor protection chiefs and department foremen. The questionnaire requested information about job history and job tasks, exposure to eight specific solvents or classes of solvents (aliphatic solvents, alicyclic solvents, toluene, xylene, benzene, chloroform, dichloromethane, and other solvents), antineoplastic agents, carcinogens, hormones, antibiotics, heavy lifting, known chronic diseases, acute diseases during pregnancy, smoking status, and previous pregnancies. Exposure frequency to each solvent was based on the cumulative weighted sum of the number of days/week the woman was exposed to the solvent. While overall response to the questionnaire was 93%, less than half the questionnaires contained information about smoking or previous pregnancies, precluding inclusion of these variables in the analysis. The distribution of broad categories of occupations (i.e., pharmaceutical workers and packers, laboratory assistants) was similar in both groups. However, exposure to each of the solvents was higher in the cases compared with controls, and the results for dichloromethane were relatively strong. For dichloromethane, the prevalence of exposure was 28.9 and 14.3% in cases and controls, respectively, resulting in an odds ratio (OR) of 2.3 (95% CI, 1.0, 5.7). There was also evidence of an increasing risk with higher exposure frequency, with an odds ratio (OR) of 2.0 (95% CI 0.6, 6.6) with exposures of less than once a week and 2.8 (95% CI 0.8, 9.5) with exposures of once a week or more. An association was also seen with exposure to four or more solvents (OR 3.4, 95% CI 1.0, 12.5), and weaker associations were seen with other specific solvents (e.g., chloroform, toluene).

Bell et al. (1991) investigated the relation between birth weight and maternal exposure to airborne dichloromethane as a result of living around the triacetate film facility in Rochester,

New York. For this population-based cross-sectional study, birth certificates were obtained for all births in 1976–1987 in Monroe County, where the triacetate film facility is located. Multiple births and births of infants weighing <750 grams were excluded. Data abstracted from the certificate included date of birth, census tract of residence, age, race, educational level of the mother and father, sex, gestational age, multiple births, month of the pregnancy that prenatal care began, total previous births, total previous live births, and conditions present during the pregnancy. An air dispersion modeling system for 250 air emissions, including dichloromethane, predicting average annual ground level concentrations in the surrounding community, was used to assign dichloromethane exposure levels to each birth mother. One of four levels of exposure was assigned to each census tract based on the isopleth of exposure in which more than half of the census tract population resided. Because of the few births among nonwhites that occurred in areas of higher exposure, the study was restricted to whites (n = 91,302). The number of births that occurred in each of the four exposure levels was n = 1,085 in the high-exposure group (50 $\mu g/m^3$ [0.014 ppm]), n = 1,795 in the moderate-exposure group (25 $\mu g/m^3$ [0.007 ppm]), n = 6,044 in the low-exposure group (10 $\mu g/m^3$ [0.003 ppm]), and n = 82,076 in the no-exposure group. At the levels of dichloromethane exposure in this population no significant adverse effect on birth weight was found. There was an 18.7 g decrease in birthweight (95% CI –51.6, 14.2) in the high- compared with the no-exposure group, adjusting for maternal age, maternal education, parity, previous pregnancy loss, late start of prenatal care, sex of the child, and pregnancy complications. No significant association was found between any combination of exposure levels and birth weight. There was no association between exposure group and risk of a low birthweight infant (i.e., <2,500 g, OR 1.0 [95% CI 0.81, 1.2] in the high- compared with the no-exposure group). The authors point out a number of problems with assignment of dichloromethane exposure. It is possible that the dichloromethane exposure was overestimated using the model. Comparisons to ambient air sampling levels collected six times/year resulted in the dichloromethane exposure derived from the model being twice as high as the ambient air samples. There was also inaccuracy in the assignment of dichloromethane exposure level to each birth because the exposure assignment was made using the predominant value of the isopleth for a census tract.

Two studies have investigated the occurrence of oligospermia among men occupationally exposed to dichloromethane exposure. Kelly (1988) studied 34 men employed in an automotive plant as bonders, finishers, and press operators. These men were self-referred to a health center for a variety of complaints, including neurological symptoms, musculoskeletal symptoms, and shortness of breath. Twenty-six of the men were bonders and eight were finishers or press operators. The job as bonder consisted of dipping hands into an open bucket of dichloromethane and splashing it onto plastic automobile parts. The dichloromethane exposure for bonders averaged 68 ppm with a range from 3.3 to 154.4 ppm. Eight men, all of whom were bonders, reported symptoms of testicular and epidydimal tenderness, with confirmation on medical exam.

They ranged in age from 20 to 47 years old and had been bonders for up to 2.9 years. The COHb levels for the eight workers with genital symptoms ranged from 1.2 to 17.3%, with an average of 6.9% anywhere from 4 to 90 hours postexposure. The COHb levels for the two men who smoked were among the highest, namely 7.3 and 17.3%. Four of the eight workers agreed to provide semen samples; their sperm counts were 2–26 $\times 10^6/cm^3$. The authors stated that men with sperm counts as low as 25 $\times 10^6/cm^3$ may still be fertile, but none of these men had had any children since working with dichloromethane despite not using contraceptives. There was one miscarriage. All four men reported dipping their hands into open buckets of dichloromethane without any protective equipment, and two men reported feeling dizzy, giddy, and high at work.

Based on the results of the Kelly (1988) case report, Wells et al. (1989) planned to do a study of oligospermia among 20 exposed and 20 unexposed to dichloromethane. The exposed workers were unvasectomized men who had worked for the 3 months prior to recruitment in furniture stripping shops. Eleven men were recruited from among 14 eligible workers at six different shops where dichloromethane was utilized. Names of acquaintances of the exposed were solicited as potential referents. Only one exposed man provided any names. Therefore, the study was redirected as a case report on the 11 exposed men. The mean TWA dichloromethane exposure was 122 ppm (range 15–366 ppm) with a mean COHb of 5.8% (range 2.2–13.5%). The mean COHb for smokers, 10.2% (range 8.1–13.5), was higher than for nonsmokers, 3.9% (range 2.2–5.9), and the nonsmoker levels were higher than the 2% level considered to be the upper limit of normal in nonsmoking populations. The mean sperm count was 54 $\times 10^6/cm^3$ (range 23–128 $\times 10^6/cm^3$) compared to a population value of 47 $\times 10^6/cm^3$ for the same geographic based on samples analyzed at the same laboratory. Using the standard definition for oligospermia of 20 $\times 10^6/cm^3$, none of the 11 workers had oligospermia.

4.1.2.9. *Summary of Noncancer Studies*

The clinical and workplace studies of noncancer health effects of dichloromethane exposure have examined markers of disease and specific clinical endpoints relating to cardiac, neurological disease, hepatic function, and reproductive health.

Cardiac effects

The effect of dichloromethane on the formation of COHb (Stewart et al., 1972b) raised concerns about potential risk of cardiovascular damage. To date, there is little evidence of cardiac damage related to dichloromethane exposure in the cohort studies of dichloromethane-exposed workers that examined ischemic heart disease mortality risk (Hearne and Pifer, 1999; Tomenson et al., 1997; Gibbs et al., 1996; Lanes et al., 1993) or in two small cardiac monitoring studies (Ott et al., 1983d; Cherry et al., 1981). However, limitations in these studies should be noted, including the healthy worker effect and the absence of data pertaining to workers who died before the establishment of the analytic cohort (Gibbs et al., 1996; Gibbs, 1992).

Neurological effects

The acute effects of dichloromethane exposure on neurological function seen in numerous case reports have also been established in experimental studies in humans (Putz et al., 1979; Winneke, 1974; Stewart et al., 1972a, b). Relatively less is known about the long-term effects of chronic exposures in humans. Some data from studies of workers suggest that the effects of dichloromethane are relatively short-lived. For example, in the study by Cherry et al. (1983) of 56 exposed and 36 unexposed workers, alterations in mood or in digit substitution test results were seen during the course of a work shift but were not seen at the beginning of a shift. No difference in four neurological symptoms was seen in an analysis of exposed workers (average exposure 475 ppm, ≥10-year duration) and an unexposed comparison group by Soden (1993). Other data suggest an increase in prevalence of neurological symptoms among workers (Cherry et al., 1981) and possible detriments in attention and reaction time in complex tasks among retired workers (Lash et al., 1991). These latter two studies are limited by the small sample size. Thus Cherry et al. (1981) and Lash et al. (1991) have low power for detecting statistically significant results and consequently should not be interpreted as definitive analyses showing no effects. Rather, these analyses provide some evidence of an increased prevalence of neurological symptoms among workers with average exposures of 75–100 ppm (Cherry et al., 1981) and long-term effects on specific neurological measures (i.e., attention and reaction time) in workers whose past exposures, at least for part of their work history, were in the 100–200 ppm range (Lash et al., 1991). The increased risk of suicide (approximately a twofold increased risk) seen in two of the worker cohort studies (Hearne and Pifer, 1999; Gibbs, 1992) is an additional indication of potential neurological consequences of dichloromethane exposure. Adequate studies addressing these specific issues are not available. Thus, given the suggestions from the currently available studies, the statement that there are no long-term neurological effects of chronic exposures to dichloromethane cannot be made with confidence.

Hepatic effects

Three studies provide data pertaining to markers of hepatic damage (i.e., serum enzymes and bilirubin levels) (Soden, 1993; Kolodner et al., 1990; Ott et al., 1983c). Two of these studies were based in the Rock Hill, South Carolina, cellulose triacetate fiber plant (Soden, 1993; Ott et al., 1983c), with the most recent of the studies focusing on workers with more than 10 years duration in a high exposure area (average exposure estimated as 475 ppm). There is some evidence of increasing levels of serum bilirubin with increasing dichloromethane exposure in Ott et al. (1983c) and Kolodner et al. (1990), but there are no consistent patterns with respect to the other hepatic enzymes examined (serum γ-glutamyl transferase, serum AST, serum ALT). These studies do not provide clear evidence of hepatic damage in dichloromethane-exposed workers, to the extent that this damage could be detected by these serologic measures; however, these data

are limited and thus the absence, presence, or extent of hepatic damage is not known with certainty.

Immune effects

Only limited, and somewhat indirect, evidence pertaining to immune-related effects of dichloromethane in humans is available. No risk was seen in the broad category of infectious and parasite-related mortality reported by Hearne and Pifer (1999), but there was some evidence of an increased risk for influenza and pneumonia-related mortality at two cellulose triacetate fiber production work sites in Maryland and South Carolina (Gibbs, 1992). In the Maryland facility, an increased risk of cervical cancer was seen among the 938 female workers, with an SMR of 3.0 (95% CI, 0.96, 6.9) in the 50–100 ppm group and 5.4 (95% CI 0.13, 30.1) in the 350–700 ppm group (Gibbs et al., 1996). Cervical cancer is viral mediated (human papilloma virus), and immunosuppression is a risk factor for development of this disease, as seen by the increased risk in immunocompromised patients and people taking immunosuppressant medications (Leitao et al., 2008; Ognenovski et al., 2004).

Reproductive effects

Studies pertaining to various reproductive effects and dichloromethane exposure from workplace settings (Wells et al., 1989; Kelly, 1988; Taskinen et al., 1986) or environmental settings (Bell et al., 1991) have examined possible associations with spontaneous abortion (Taskinen et al., 1986), low birth weight (Bell et al., 1991), and oligospermia (Wells et al., 1989; Kelly, 1988). Of these, the data pertaining to spontaneous abortion provide the strongest evidence of an adverse effect of dichloromethane exposure, particularly with respect to the case-control study in which the strongest association was seen specifically with the higher frequency category of dichloromethane exposure. It is a small study (44 cases, 130 controls), however, with limited quantitative exposure assessment and multiple exposures (although the association seen with dichloromethane was among the highest seen among the solvents) and so cannot be considered to firmly establish the role of dichloromethane in induction of miscarriage. However, the high exposure scenario, including the potential for substantial dermal exposure in the study of Kelly (1988), also suggests the potential for adverse male reproductive effects.

4.1.3. Cancer Studies

4.1.3.1. *Identification and Selection of Studies for Evaluation of Cancer Risk*

Twelve epidemiologic studies of cancer risk were identified and included in this evaluation: four cohorts for which the primary solvent exposure was to dichloromethane (two in film production settings and two in cellulose triacetate fiber production), one large cohort of civilian employees at a military base with exposures to a variety of solvents but that included an assessment specifically of dichloromethane exposure, and seven case-control studies of specific

cancers with data on dichloromethane exposure. One additional study (Ott et al., 1985), a cohort of 1,919 men employed at Dow Chemical facilities, was identified but was not included in the summary. The analysis was based on exposure to a combined group of chlorinated methanes (e.g., carbon tetrachloride, chloroform, methyl chloride, and dichloromethane), and it was not possible from the data presented to assess the individual effects of dichloromethane.

4.1.3.2. *Description of the Selected Studies*

In this section, the study setting, methods (including exposure assessment techniques), results pertaining to incidence or mortality from specific cancers, and a brief summary of primary strengths and limitations are summarized for each of the 12 selected studies. When two papers of the same cohort were available, the results from the longer period of follow-up are emphasized in the summary. Information from earlier reports is used when these reports contain more details regarding working conditions, study design, and exposure assessment. The description of individual studies is followed by a summary of the evidence available from these studies relating to specific types of cancer.

4.1.3.3. *Cellulose Triacetate Film Base Production Cohorts*

4.1.3.3.1. *Cellulose triacetate film base production—Rochester, New York (Eastman Kodak).*

Friedlander et al. (1978) reported a cohort mortality study of workers in an Eastman Kodak facility in Rochester, New York. This study was expanded and extended several times during the next 20 years (Hearne and Pifer, 1999; Hearne et al., 1990, 1987). The latest analysis provided data on two overlapping cohorts. The first cohort (Cohort 1) consisted of 1,311 male workers employed in the roll coating division (n = 1,070) or the dope and distilling departments (n = 241) of the Eastman Kodak facility in Rochester, New York. Men who began working in these areas after 1945 and were employed in these areas for at least 1 year (including seasonal or part-time work that equaled 1 full-time year equivalent) from 1946 to 1970 were included. Follow-up time was calculated from the end of the first year of employment in the study area through December 31, 1994. The mean duration of work in Cohort 1 was 17 years. The total number of person-years of follow-up was 46,112, and the mean duration of follow-up was 35.2 years (range 25–49 years). The second cohort (Cohort 2) included 1,013 male workers in the roll coating division who were employed for at least 1 year in this division between 1964 and 1970. Follow-up time was calculated from January 1, 1964, or the date of first employment in the roll coating division for those who were employed there before 1964 and those who began in 1964 or later, respectively. Follow-up continued through December 31, 1994. The mean duration of work in Cohort 2 was 24 years. Total follow-up time was 26,251 person-years, and the mean duration of follow-up was 25.9 years (range 25–31 years). Cohort 2 was the focus of previous analyses by Friedlander et al. (1978) and Hearne et al. (1990, 1987).

For both cohorts, causes of death were based on the underlying causes of death recorded on the death certificates, which were routinely obtained by the company for the processing of life insurance claims. The expected number of deaths was calculated using appropriate age-, sex-, calendar time-, and cause-specific death rates for men in New York State (excluding New York City). In addition, another referent group was also used in the analysis of the second cohort. This other referent was based on the age-, sex-, calendar time-, and cause-specific death rates of other hourly male workers employed at the Eastman Kodak plant in Rochester, New York. (An internal referent group was also described for Cohort 1, but data for that analysis were not presented.)

Dichloromethane was first used in the film production process at the Eastman Kodak facility around 1944 (Hearne et al., 1987). Cellulose triacetate was dissolved in dichloromethane and then cast into a thin film onto revolving wheels. The film was then cured by circulating hot air in the coating machines, and the solvent was recovered and redistilled. 1,2-Dichloropropane and 1,2-dichloroethane were also used as solvents from the 1930s to the 1960s, but dichloromethane was predominant (ratio 17:2:1 for dichloromethane:1,2-dichloropropane:1,2-dichloroethane in general workplace air measurements) (Hearne et al., 1987).

The exposure assessment in the Rochester, New York, Eastman Kodak cohort studies was based on employment records (start and stop dates for specific jobs in the relevant areas of the company) in combination with air monitoring data used to estimate the exposure level for a given job, location, and time period (Hearne et al., 1987). Air monitoring began in the 1940s, but few data are available before 1959. In the most recent update (Hearne and Pifer, 1999), more than 1,500 area and 2,500 breathing zone air samples were used in the exposure assessment process. Reductions in exposures in the dope department and the distilling department began after 1965. The highest exposure jobs were operator and maintenance workers (dope department) and filter washing and waste operator (distilling department), with estimated 8-hour TWA exposures of 100–520 ppm between 1946 and 1985. There was little change in estimated exposures for jobs in the roll coating division from the 1940s through 1985, but some reduction was seen from 1986 to 1994. The mean 8-hour TWA exposures were 39 ppm for Cohort 1 and 26 ppm for Cohort 2.

These data were used to estimate a cumulative exposure index (i.e., the summation across all jobs held by an individual of the product of the average dichloromethane concentration as ppm and the duration of employment in that job). The authors refer to this as a "career exposure index." Additional adjustment in these estimates was made for respiratory protection, but the details of this adjustment were not described. For Cohort 1, the cumulative exposure categories used in exposure-effect analyses were <150, 150–349, 350–799, and ≥800 ppm. For Cohort 2, the cumulative exposure categories were <400, 401–799, 800–1,199, and ≥1,200 ppm. The cut points were chosen to produce an approximately equal number of expected total deaths in these categories. Two different methods to calculate expected number of deaths within each exposure

category were used for each cohort analysis. For Cohort 1, an internal comparison was made based on the distribution of person-years within each exposure category, and an external comparison was made applying New York State mortality rates. For Cohort 2, the internal comparison using the distribution of person-years within each exposure category was also used, but the external comparison was based on mortality rates in other hourly workers at the Rochester, New York, Eastman Kodak work site.

There was no increased risk of mortality for all sites of cancer or for lung cancer in either cohort analysis (Table 4-4). Data pertaining to smoking history, obtained from a survey of workers in the New York film production facility, indicate that smoking rates were similar in the exposed group, the internal comparison group, and the general population; therefore, it is unlikely that differences in smoking could be masking an effect of dichloromethane.

The only specific sites for which there were increased SMRs in both cohorts were brain and CNS cancer, Hodgkin's lymphoma, and leukemia. Pancreatic cancer mortality risk was increased in Cohort 2 but not in Cohort 1. None of these associations were statistically significant, and the Hodgkin's lymphoma observations were based on a total of only four cases in both cohorts combined and so were very imprecise. Within Cohort 2, there was little difference in results for most sites, using the different referent groups, but the point estimates for the SMRs for brain and CNS cancer, Hodgkin's lymphoma, and leukemia were somewhat higher using the New York State referent group compared with the internal Eastman Kodak referent group. An attenuation of the dichloromethane association seen in the analyses using the internal Kodak referent group would be expected if the risk of specific cancers was increased in this comparison group, possibly because of other workplace exposures.

Table 4-4. Mortality risk in Eastman Kodak cellulose triacetate film base production workers, Rochester, New York

	Cohort 1: 1,311 men employed 1946–1970, followed through 1994				Cohort 2: 1,013 men employed 1964–1970, followed through 1994						
	New York referent group				New York referent group				Kodak referent group		
Cancer type	Obs[b]	Exp[b]	SMR	95% CI	Obs	Exp	SMR	95% CI	Exp	SMR	95% CI
Cancer, all sites	93	105.8	0.88	0.71–1.08	91	102.0	0.89	0.72–1.10	94.7	0.96	0.77–1.18
Liver[a]	1	2.4	0.42	0.01–2.36	1	2.4	0.42	0.01–2.33	1.8	0.55	0.01–3.07
Pancreas	5	5.5	0.92	0.30–2.14	8	5.3	1.51	0.65–2.98	5.1	1.55	0.67–3.06
Lung[a]	27	36.0	0.75	0.49–1.09	28	34.2	0.82	0.55–1.19	31.3	0.89	0.59–1.29
Brain[a]	6	2.8	2.16	0.79–4.69	4	2.1	1.88	0.51–4.81	2.7	1.46	0.39–3.75
Lymphatic system	5	6.6	0.75	0.24–1.78	6	5.7	1.06	0.39–2.30	5.7	1.05	0.38–2.28
Non-Hodgkin's	2	4.1	0.49	0.06–1.76	3	3.5	0.85	0.17–2.50	3.6	0.84	0.17–2.46
Hodgkin's	2	1.1	1.82	0.20–6.57	2	0.6	3.13	0.35–11.30	0.9	2.23	0.25–8.05
Multiple myeloma	1	1.5	0.68	0.01–3.79	1	1.5	0.65	0.01–3.62	1.3	0.79	0.01–4.39
Leukemia	8	3.9	2.04	0.88–4.03	6	3.5	1.73	0.63–3.76	4.4	1.37	0.50–2.98

[a]Liver includes liver and biliary duct; lung includes lung, trachea, and bronchus; brain includes brain and CNS.
[b]Obs = number observed deaths, Exp = number of expected deaths.

Source: Hearne and Pifer (1999).

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The authors presented the exposure-effect analysis, based on the estimated cumulative dichloromethane exposure groups, for all sites of cancer, pancreatic cancer, lung cancer, brain cancer, and leukemia (Table 4-5).

Table 4-5. Mortality risk by cumulative exposure in Eastman Kodak cellulose triacetate film base production workers, Rochester, New York

Cohort, cancer, referent group	SMRs (number of observed deaths)				
		Cohort 1[a]			
Cumulative exposure (ppm years)	<150	150–349	350–799		≥800
Cancer, all sites					
internal	0.81 (20)	1.02 (19)	1.10 (28)		1.07 (26)
New York	0.67	0.93	0.95		1.00
Pancreas					
internal	0.74 (1)	0.00 (0)	0.77 (1)		2.34 (3)
New York	0.68	0.00	0.65		2.18
Lung[b]					
internal	0.78 (5)	1.07 (6)	1.25 (9)		0.90 (7)
New York	0.52	0.90	0.86		0.77
Brain[b]					
internal	0.58 (1)	0.78 (1)	1.65 (3)		0.85 (1)
New York	1.10	1.77	3.99		1.78
Leukemia					
internal	0.83 (2)	0.00 (0)	0.48 (1)		2.73 (5)
New York	1.61	0.00	0.98		5.79
		Cohort 2[c]			
Cumulative exposure (ppm years)		<400	400–799	800–1,199	≥1,200
Cancer, all sites					
Internal		0.89 (18)	0.96 (33)	1.11 (23)	1.08 (17)
New York		0.76	0.93	1.13	1.12
Pancreas					
Internal		2.58 (4)	0.00 (0)	0.95 (2)	1.43 (2)
New York		2.86	0.00	1.83	2.67
Lung[b]					
Internal		0.95 (6)	1.15 (12)	0.94 (6)	0.82 (4)
New York		0.80	1.00	0.89	0.79
Brain[b]					
Internal		0.00 (0)	1.13 (2)	1.37 (1)	1.49 (1)
New York		0.00	2.02	1.75	2.50
Leukemia					
Internal		0.00 (0)	0.84 (2)	0.75 (1)	2.70 (3)
New York		0.00	1.26	1.10	4.84

[a]Cohort 1: 1,311 men employed 1946–1970 in the roll coating division, dope department, or distilling department, followed through 1994; mean exposure (cumulative exposure years) 66, 244, 543, and 1,782 ppm-years in the four dose groups, respectively.
[b]Lung includes lung, trachea, and bronchus; brain includes brain and CNS.
[c]Cohort 2: 1,013 men employed 1964–1970 in the roll coating division, followed through 1994; mean exposure (cumulative exposure years) 168, 581, 981, and 1,670 ppm-years in the four dose groups, respectively.

Source: Hearne and Pifer (1999).

There is no evidence of an exposure-effect for all site cancer mortality or lung cancer mortality risk. The relatively sparse number of deaths for the other specific cancer types makes it difficult to interpret the data. The patterns for pancreatic cancer differ between the two cohorts, with increased risk at the higher dose in Cohort 1 and a U-shaped curve seen in Cohort 2. For brain cancer mortality, a higher SMR was seen in the groups with cumulative exposure levels of 800 ppm-years or greater compared with lower exposure groups. For leukemia, in both cohorts, an increased mortality risk is seen in the highest exposure group (mean approximately 1,700 ppm-years).

A strength of the Eastman Kodak cohort studies was the sampling data for dichloromethane that allowed an assessment of each worker's exposure, using the monitoring data and the worker's job history, making exposure-effect analyses possible. Follow-up of the vital status of the cohort was >99% (Hearne and Pifer, 1999). There was also some information on smoking history, too, for workers in the plant, based on a survey conducted in 1986 (Hearne et al., 1987). A difficulty in interpreting the data, however, is that there was some overlap between the cohorts: 707 of the men were included in both Cohort 1 and Cohort 2. Data are not presented in a way that would allow the reader to eliminate duplicate cases and person-years so that cases are only counted once when examining both cohorts. A strength of the Cohort 1 sampling strategy, compared with that of Cohort 2, is that Cohort 1 is limited to workers who began work at the plant after 1945. These workers would not have had workplace exposure to methanol and acetone, which were used at the plant in the film production process prior to that time. Also, follow-up began with the beginning of employment in the relevant area. In contrast, Cohort 2 was limited to workers who were employed from 1964 to 1970, so exposed workers who left or died before 1964 were not included. The relatively small number of cases with specific low incidence cancers (e.g., brain, leukemia) is also a limitation of the analyses of both of the cohorts in this study. In addition, the exposure levels in both cohorts (mean 8-hour TWA 39 and 26 ppm in Cohorts 1 and 2, respectively) is relatively low compared with values seen in other workplaces, including the cellulose triacetate fiber production cohorts described in Ott et al. (1983a) and Gibbs et al. (1996). Also, the outcome assessment is based on mortality (underlying cause from death certificates) rather than incidence data, and, because the Kodak studies were limited to men, there is no information on risk of breast cancer or other female reproductive cancers.

4.1.3.3.2. ***Cellulose triacetate film base production—Brantham, United Kingdom (Imperial Chemical Industries).*** Tomenson et al. (1997) reported the results of a retrospective cohort mortality study of 1,473 men who worked at a film-base production facility in Brantham, England, anytime between 1946 and 1988 in jobs that were considered to have dichloromethane exposure. The start of the follow-up period was not specified by the authors but is likely to have been 1946 or the date of first employment at the plant. Follow-up of the cohort continued

through December 31, 1994, and vital status was based on national records (United Kingdom National Health Service Central Register and the Department of Social Security). Causes of death were based on the underlying causes of death recorded on the death certificates. The expected number of deaths was calculated using age-, sex-, calendar time-, and cause-specific death rates for England and Wales. In addition, a comparison using mortality rates for the local areas (Tendring and Samford) for 1968–1978 and analyses limited to workers who had been employed for at least 3 months were also made, but the results of these analyses were not presented. The mean duration of work in the cohort was 9 years, the total number of person-years was 39,759, and the mean duration of follow-up was 27.0 years (7–49 years).

This facility produced cellulose diacetate film from 1950 to 1988, with other types of films also manufactured beginning in the 1960s. Dichloromethane was the solvent used in this process, and exposure occurred in the production of the triacetate film base and the casting of the film into rolls. The exposure assessment was based on more than 2,700 personal or air monitoring samples collected since 1975. An exposure matrix was constructed, assigning jobs to 1 of 20 work groups with similar exposure potential for each of four different time periods (before 1960, 1960–1969, 1970–1979, and 1980–1988). For the 1980–1988 period, exposure estimates for specific jobs were based on about 330 personal monitoring samples. For the earlier time periods, information about work tasks and location was used in combination with the information about the number of, use of, speed of, and problems with casting machines at different times from their initial introduction in 1950. The highest exposures were estimated to be in the casting machine operators and cleaners. Lifetime cumulative exposure to dichloromethane was calculated as the product of the mean level of exposure for the assigned work group and the duration of employment in that job summed across all jobs. Three categories of cumulative exposure were used for the analysis of ever exposed workers: <400, 400–700, and 800+ ppm-years. Approximately 30% of the workers in the cohort were classified as "unassigned" for the calculation of exposure group because sufficient information needed to determine exposures (i.e., the location and tasks assigned to laborers and maintenance workers) was not available. The mean 8-hour TWA exposure was estimated as 19 ppm for the cohort.

There was no increased risk of mortality for all sites of cancer (Table 4-6), and the SMRs for most of the specific cancer sites examined (stomach, colon, rectum, liver, pancreas, lung, and prostate) were less than 1.0. The only specific sites for which there was an increased SMR (i.e., 1.1 or higher) were brain and CNS cancer and leukemia, and these estimates were based on few (less than five) observed cases (Table 4-6). Tomenson et al. (1997) present the exposure-effect analysis, based on the estimated cumulative dichloromethane exposure groups, for all sites of cancer, pancreatic cancer, and lung cancer, and there is no evidence of an increasing effect with increasing exposure level in these analyses. A formal exposure-effect analysis for brain cancer or leukemia was not presented. However, the authors described two of the brain cancer cases as having "minimal" exposure to dichloromethane (and thus presumably would have been in the

<400 ppm-year cumulative exposure group). One case was estimated as having 572 ppm-years cumulative exposure, and the other case was an electrician classified in the unassigned exposure group. He had worked for 21 years at an exposure level "that was unlikely to have exceeded 15 ppm 8 hour TWA."

Table 4-6. Mortality risk in Imperial Chemical Industries cellulose triacetate film base production workers, Brantham, United Kingdom: 1,473 men employed 1946–1988, followed through 1994

Cancer type	Observed	Expected[a]	SMR	95% CI
Cancer, all sites	68	104.6	0.65	0.51–0.82
Liver and biliary duct	0	1.5	–	–
Pancreas	3	4.4	0.68	0.14–1.99
Lung, trachea, bronchus	19	41.3	0.46	0.29–0.75
Brain and CNS system	4	2.8	1.45	0.40–3.72
Lymphatic and hematopoietic	6	7.1	0.85	0.31–1.84
Leukemia	3	2.7	1.11	0.23–1.84

[a]Expected, calculated from observed and SMR data reported by the authors by using the following formula: expected = 100 × observed ÷ SMR; SMRs and CIs were not calculated for categories with zero observed cases.

Source: Tomenson et al. (1997).

A strength of this study was the monitoring data available that allowed assignment of cumulative exposure categories for use in exposure-effect analyses. However, 30% (439) of exposed workers had insufficient work histories to determine lifetime cumulative exposure. Air measurements were not available until 1975, and personal measures were not available until 1980. In addition, the duration of exposure was relatively low (mean, 9 years), the mean exposure level was relatively low (mean 8-hour TWA, 19 ppm), and there were very few deaths from specific types of cancer, which limit the statistical power of the study to examine associations among dichloromethane and specific cancers. Other limitations, as were also noted in the Kodak cohort studies, include the use of mortality rather than incidence to define risk, the reliance solely on underlying causes of death from death certificates to classify specific cancer types and the lack of information on breast cancer risk.

4.1.3.4. *Cellulose Triacetate Fiber Production Cohorts*

4.1.3.4.1. *Cellulose triacetate fiber production—Rock Hill, South Carolina (Hoechst Celanese Corporation).* Two cohorts of cellulose triacetate fiber workers have been studied in Rock Hill, South Carolina (Lanes et al., 1993, 1990; Ott et al., 1983a, b), and Cumberland, Maryland (Gibbs et al., 1996; Gibbs, 1992). Workers were exposed to dichloromethane, methanol, and acetone in both facilities.

Ott et al. (1983a, b) conducted a retrospective cohort mortality study of 1,271 acetate fiber production workers (551 men and 720 women) employed at least 3 months from 1954 to

1977 at Dow Chemical Company, Rock Hill, South Carolina. This analysis focused on ischemic heart disease mortality risk, and there was no presentation of cancer risk. The Rock Hill cohort study was updated twice, through September 30, 1986 (Lanes et al., 1990), and December 31, 1990 (Lanes et al., 1993), and analyses of cancer mortality risks were included in these later reports. Causes of death information was obtained from death certificates, with coding based on the underlying and contributing causes (Ott et al., 1983a). The referent used in the updates was the general population of York County, South Carolina, and analyses were adjusted for age, race, gender, and calendar period. Because the results of the mortality risk analyses were similar for both updates, those from the 1993 paper are presented here. The mean duration of work in the cohort was not reported, but 56% worked for fewer than 5 years (calculated from Tables 3 and 4 of Ott et al., 1983b). The mean duration of follow-up was 23.6 years in the analysis through 1986 (Lanes et al., 1990) but was not reported in the later paper (Lanes et al., 1993). The 1993 report added approximately 4.25 years of follow-up, which would result in an estimate of approximately 28 years follow-up for this report.

The Rock Hill, South Carolina, plant began producing cellulose triacetate fiber in 1954. Dichloromethane was used as the solvent for the initial mixing with cellulose triacetate flakes. This mixture was then filtered and transferred to the extrusion area for drying and winding. Air measurements taken in 1977–1978 were assumed to be representative of operations since dichloromethane use began in 1954, based on review of processing operations. The median 8-hour TWA exposures were estimated as 140, 280, and 475 ppm in the low, moderate, and high categories of exposure (Ott et al., 1983a). Employment records provided information on jobs held and dates employed, and this was used in conjunction with the exposure estimates for specific jobs and work areas to classify individual exposures. However, detailed work history information was only available for 475 (37%) of the workers (Lanes et al., 1990), and it is not clear how the exposure assessment was applied to workers with missing job history data.

Methanol was also used in the cellulose triacetate fiber production process, and methanol exposure was estimated as one-tenth that of dichloromethane. Acetone exposure was used in the production of acetate (cellulose diacetate) fiber at an adjacent part of the plant. The exposure to acetone was inversely related to that of dichloromethane, with estimated median 8-hour TWAs of 1,080 ppm acetone in the low dichloromethane exposure group and 110 ppm acetone in the moderate and high dichloromethane groups in the Rock Hill plant (Ott et al., 1983a).

In the latest follow-up (Lanes et al., 1993), there was no increase in mortality risk from cancer (all sites) or from cancer of the lung or pancreas (Table 4-7). The SMR for liver and bile duct cancer, based on four observed cases, was 2.98 (95% CI 0.81, 7.63). This was lower than the SMR of 5.75 (95% CI 1.82, 13.8) that was reported in the 1990 analysis, based on these same four cases but on a shorter follow-up period (and thus lower number of expected cases). Three of these cases were bile duct cancers. This was the first cohort study that included women, and this study provided data on breast cancer risk. There were 3 observed breast cancer deaths

compared with 5.59 expected, yielding an SMR of 0.54 (95% CI 0.11, 1.57). No data were provided pertaining to reproductive risk factors (e.g., pregnancy history) for breast cancer among the women in this cohort, so it is difficult to assess whether these potential confounders are likely to have been distributed differently in the cohort compared with the referent group. Information about brain cancer, Hodgkin's lymphoma, and leukemia (Table 4-7) was not included in this report but was included in the report by Gibbs (1992) (see Table 11 of that report).

Table 4-7. Mortality risk in Hoechst Celanese Corporation cellulose triacetate fiber production workers, Rock Hill, South Carolina: 1,271 men and women employed 1954–1977, followed through 1990

Cancer type	Observed	Expected	SMR[b]	95% CI[b, c]
Cancer, all sites	39	47.7	0.82	0.58–1.52
Liver and biliary duct	4	1.34	2.98	0.81–7.63
Pancreas	2	2.42	0.83	0.10–2.99
Lung, trachea, bronchus	13	16.21	0.80	0.43–1.37
Brain and CNS[a]	1	1.5	0.67	0.2–3.71
Hodgkin's lymphoma[a]	0	0.24	–	–
Leukemia[a]	1	1.11	0.90	0.02–5.0
Breast cancer (women)	3	5.59	0.54	0.11–1.57

[a]Data for brain and CNS cancer, Hodgkin's lymphoma, and leukemia were reported in Gibbs (1992).
[b]SMRs and CIs were not calculated for categories with zero observed cases.
[c]CIs were calculated from Breslow and Day (1987, Table 2.10).

Source: Lanes et al. (1993).

There are a number of limitations of this study, including the small size of the cohort, small number of observed cancer deaths, availability of detailed work history information for only 37% of the workers, and use of mortality rather than incidence data. The exposure levels at this plant were high, but the duration of exposure for most of the cohort was relatively short (<5 years). It is the first cohort study, however, that included women and provided information on breast cancer risk.

4.1.3.4.2. ***Cellulose triacetate fiber production—Cumberland, Maryland (Hoechst Celanese Corporation).*** Gibbs et al. (1996) studied a cohort of 2,909 cellulose triacetate fiber production workers (1,931 men and 978 women) at a Hoechst Celanese plant in Cumberland, Maryland. This retrospective cohort mortality study included all workers who were employed on or after January 1, 1970, and who worked at least 3 months. This study also included a very small comparison group (256 men, 46 women) that was described as a "0" or "no" exposure group of workers at the plant who worked in jobs that were not considered to have had dichloromethane exposure, for a total of 2,187 men and 1,024 women in the exposed and nonexposed groups.

The plant closed in 1981, and mortality was followed through 1989. Since 1955, employees of this plant were exposed to dichloromethane, methanol, acetone, and finishing oils used as lubricants. Before 1955, acetone was the only exposure. Industrial hygiene monitoring focusing on dichloromethane, acetone, and methanol began in the late 1960s. Exposure groupings (low, 50–100 ppm, and high, 350–700 ppm) were assigned by area in which employees worked. The extrusion and spinning workers and jet wipers were among the high exposure group (300–1,250 ppm 8-hour TWA). The SMR analysis that was reported used Allegany County, Maryland, as the comparison group. Cause of death information was obtained from death certificates, but the authors did not state whether they used underlying or underlying and contributing cause of death information. The mean duration of work in the cohort was not reported. The total follow-up period included 49,828 person-years (16,292 in the high exposure group and 33,536 in the low exposure group), and the mean duration of follow-up was 17.2 years (range 8–20 years). These data were found in Hearne and Pifer (1999, Table 7).

There was little evidence of an increase in mortality risk from cancer (all sites) or from cancer of the liver and bile duct, pancreas, or brain in men or in women (Table 4-8). An increasing risk with increasing exposure level was seen for prostate cancer mortality in men. The *p*-value for the trend was not given, but the authors describe it as a "nonstatistically significant dose-response relationship." A statistically significant SMR for prostate cancer death was seen in the 350–700 ppm group when latency (at least 20 years since first exposure) was included in the analysis (SMR = 2.08, $p < 0.05$). Cervical cancer mortality risk was increased, but the small number of cases in the high exposure group did not allow a precise assessment of the pattern with respect to exposure level. There was no increased risk of breast cancer.

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Table 4-8. Cancer mortality risk in Hoechst Celanese Corporation cellulose triacetate fiber production workers, Cumberland, Maryland: 2,909 men and women employed 1970–1981, followed through 1989

Cancer type, exposure level[a]	Men (n = 1,931)				Women (n = 978)			
	Obs[b]	Exp[b]	SMR	95% CI[c]	Obs[b]	Exp[b]	SMR	95% CI[c]
Cancer, all sites	121				42			
50–100 ppm	64	70.0	0.91	0.70–1.2	37	44.79	0.83	0.58–1.1
350–700 ppm	57	75.6	0.75	0.57–0.98	5	4.61	1.1	0.35–2.5
Liver	2				0			
50–100 ppm	1	1.33	0.75	0.02–4.2	0	1.04		–
350–700 ppm	1	1.24	0.81	0.02–4.5	0	0.10		–
Pancreas	3				1			
50–100 ppm	2	2.24	0.89	0.1–3.2	1	1.73	0.58	0.01–3.2
350–700 ppm	1	2.90	0.35	0.01–1.9	0	0.18		–
Lung	35				11			
50–100 ppm	20	25.7	0.78	0.48–1.2	9	8.24	1.1	0.50–2.1
350–700 ppm	15	27.3	0.55	0.31–0.91	2	0.87	2.3	0.28–8.3
Brain[a]	2				2			
50–100 ppm	1	1.88	0.53	0.01–2.96	2	0.66	3.1	0.37–10.9
350–700 ppm	1	1.94	0.52	0.01–2.87	0	0.07		
Hodgkin's[a]								
50–100 ppm	1	0.4	2.5	0.06–13.9	0	0.23		
350–700 ppm	0	0.41			0	0.02		
Leukemia[a]								
50–100 ppm	4	2.14	1.9	0.51–4.8	0	1.25		
350–700 ppm	1	2.28	0.44	0.01–2.4	0	0.13		
Prostate	22				Not applicable			
50–100 ppm	9	6.41	1.4	0.64–2.7				
350–700 ppm	13	7.26	1.8	0.95–3.1				
Cervical	Not applicable				6			
50–100 ppm					5	1.69	3.0	0.96–6.9
350–700 ppm					1	0.19	5.4	0.13–30.1
Breast[a]	0				10			
50–100 ppm	0	0.03			9	9.8	0.92	0.42–1.7
350–700 ppm	0	0.02			1	1.07	0.93	0.02–5.2

[a]Data for brain and CNS cancer, Hodgkin's lymphoma, leukemia, and breast cancer reported in Gibbs (1992).
[b]Obs = number of observed deaths, Exp = number of expected deaths. Referent group = Allegany County, Maryland. SMRs and CIs were not calculated for categories with zero observed cases.
[c]CIs were calculated from Breslow and Day (1987, Table 2.10).

Sources: Gibbs et al. (1996); Gibbs (1992).

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2960
2961 A primary limitation of this study is that workers who were exposed before 1970 but
2962 were not working at the plant in 1970 were not included in the cohort. The authors had
2963 attempted to create a cohort of all workers who were employed on or after January 1, 1954, but
2964 problems with the completeness of the personnel file made it impossible to use this study design.

From what the author (Gibbs, 1992) was able to determine, the records of workers who had died, left the company, or retired before the mid to late 1960s (when a new personnel system was developed) were not available. Additional limitations include the small size of the cohort, small number of observed cancer deaths, and use of mortality (death certificate) data. This is particularly problematic for cancers with relatively high survival rates (such as prostate cancer and cervical cancer), since incidence rates are not estimated well by mortality rates in this situation.

4.1.3.5. *Solvent-Exposed Workers—Hill Air Force Base, Utah*

Spirtas et al. (1991) and Blair et al. (1998) evaluated exposure to dichloromethane in relation to mortality risk in successive retrospective cohort studies of 14,457 civilian workers employed at Hill Air Force Base in Utah for at least 1 year from 1952–1956. The analysis was limited to the workers that were white or who had missing data on race, resulting in a sample size of 14,066 (10,461 men, 3,605 women). Spirtas et al. (1991) examined mortality through 1982 (3,832 deaths), and Blair et al. (1998) updated mortality through 1990 (4,195 deaths). The underlying and contributing causes of death information from death certificates was used to classify cause-specific mortality. SMRs were calculated by using mortality rates from the Utah population, and an internally standardized life table method was used to adjust for age at entry into the cohort and competing causes of death. In the Blair et al. (1998) analysis, adjusted relative risks (rate ratios) were estimated from a Poisson regression analysis with unexposed workers as the referent. The mean duration of work was not reported. In the analysis through 1982 (Spirtas et al., 1991), there were 22,770 person-years of follow-up in men and 3,091 person-years of follow-up in women who were classified as exposed to dichloromethane. The total number of workers classified as exposed to dichloromethane was 1,222 (Stewart et al., 1991), which would yield an estimated mean of approximately 21 years follow-up through 1982. The total number of person-years included in the later report (Blair et al., 1998), with the addition of 8 more years of follow-up, was not reported but would be expected to increase the mean follow-up time to approximately 29 years.

Two industrial hygienists developed the exposure assessment based on walkthrough surveys, interviews with management and labor representatives, review of historical records, job descriptions, monitoring data and other information pertaining to chemicals used, and organization of the work site (Blair et al., 1998; Spirtas et al., 1991). Each worker was assigned exposure by using information on the worker's job history, which included job titles, department codes, and dates of employment. The most detailed exposure assessment was done for trichloroethylene, the primary focus of the study. Dichloromethane, one of 25 other exposures analyzed, was classified as a dichotomous exposure (ever exposed, never exposed).

Blair et al. (1998) presented the mortality risk for three specific cancers in relation to 15 of the 25 chemicals classified as dichotomized exposures. The rate ratios for non-Hodgkin's

lymphoma and multiple myeloma in relation to dichloromethane in men were 3.0 (95% CI 0.9, 10.0) and 3.4 (95% CI 0.9, 13.2), respectively. These rate ratios, (particularly those for multiple myeloma), were considerably higher than the rate ratios for any of the other chemicals examined, in which the next highest observed rate ratio was 1.8 for Freon. No cases of either of these cancers were observed in women with dichloromethane exposure, but the rate ratio for breast cancer in these women was 3.0 (95% CI 1.0–8.8). Associations of similar magnitude (rate ratios of 3.0–4.0) were also seen among breast cancer and some other exposures (Freon, solder flux, isopropyl alcohol, and trichloroethane).

This is the largest of the cohort studies that were identified that included women and specifically reported data pertaining to breast cancer risk. The major limitation of this study is that the exposure assessment for dichloromethane was based on a dichotomized classification. In addition, exposure to many different types of solvents was common; thus, it is difficult to completely separate the effects of individual exposures. Some aspects of reproductive history, such as age at first pregnancy, are known risk factors for breast cancer. Reproductive history was not included in this analysis, but Blair et al. (1998) note that it is unlikely that these factors would confound the results of a few specific chemicals, since the referent group was an internal group within the cohort (and thus would be expected to be similar in terms of socioeconomic status) and there was no association overall between solvent exposures and breast cancer mortality.

4.1.3.6. *Case-Control Studies of Specific Cancers and Dichloromethane*

Seven site-specific cancer case-control studies included dichloromethane as an exposure of interest. These studies involve six cancer sites: brain and CNS (Cocco et al., 1999; Heineman et al., 1994), breast (Cantor et al., 1995), kidney (Dosemeci et al., 1999), pancreas (Kernan et al., 1999), rectum (Dumas et al., 2000), and childhood leukemia (Infante-Rivard et al., 2005). A synopsis of cohort studies in humans is provided in Table 4-9.

4.1.3.6.1. ***Case-control studies of brain cancer.*** Heineman et al. (1994) studied the association between astrocytic brain cancer (International Classification of Diseases 9th ed. [ICD-9] codes 191, 192, 225, and 239.7) and occupational exposure to chlorinated aliphatic hydrocarbons. Cases were identified by using death certificates from southern Louisiana, northern New Jersey, and the Philadelphia area. This analysis was limited to white males who died between 1978 and 1981. Controls were randomly selected from the death certificates of white males who died of causes other than brain tumors, cerebrovascular disease, epilepsy, suicide, and homicide. The controls were frequency matched to cases by age, year of death, and study area.

Next of kin were successfully located for interview for 654 cases and 612 controls, which represents 88 and 83% of the identified cases and controls, respectively. Interviews were completed for 483 cases (74%) and 386 controls (63%). There were 300 cases of astrocytic

brain cancer (including astrocytoma, glioblastoma, mixed glioma with astrocytic cells). The ascertainment of type of cancer was based on review of hospital records, which included pathology reports for 229 cases and computerized tomography reports for 71 cases. After the exclusion of 66 controls with a possible association between cause of death and occupational exposure to chlorinated aliphatic hydrocarbons (some types of cancer, cirrhosis of the liver), the final analytic sample consisted of 300 cases and 320 controls.

In the next-of-kin interviews, the work history included information about each job held since the case (or control) was 15 years old (job title, description of tasks, name and location of company, kinds of products, employment dates, and hours worked per week). Occupation and industry were coded based on four-digit Standard Industrial Classification and Standard Occupational Classification (Department of Commerce) codes. The investigators developed matrices linked to jobs with likely exposure to dichloromethane, five other chlorinated aliphatic hydrocarbons (carbon tetrachloride, chloroform, methyl chloroform, tetrachloroethylene, and trichloroethylene), and organic solvents (Gomez et al., 1994). This assessment was done blinded to case-control status. Exposure was defined as the probability of exposure to a substance (the highest probability score for that substance among all jobs), duration of employment in the exposed occupation and industry, specific exposure intensity categories, average intensity score (the three-level semiquantitative exposure concentration assigned to each job multiplied by duration of employment in the job, summed across all jobs), and cumulative exposure score (weighted sum of years in all exposed jobs with weights based on the square of exposure intensity [1, 2, 3] assigned to each job). Secular trends in the use of specific chemicals were considered in the assignment of exposure potential. Exposures were lagged 10 or 20 years to account for latency. Thus, this exposure assessment procedure was quite detailed.

Adjusting for age and study area, the OR for the association between any exposure to dichloromethane and risk of astrocytic brain cancer was 1.3 (95% CI 0.9, 1.8). There was a statistically significant trend ($p < 0.05$) with increasing probability of exposure to dichloromethane with an OR = 1.0 (95% CI, 0.7, 1.6) for low probability, OR = 1.6 (95% CI 0.8, 3.0) for medium probability, and OR = 2.4 (95% CI 1.0, 5.9) for high probability compared with the referent group of unexposed men. An increased risk with higher duration of exposure was also observed, with OR = 1.7 (95% CI 0.9, 3.6) for 21 or more years of work in exposed jobs for all exposed workers and OR = 6.1 (95% CI 1.1, 43.8) for the combination of 21 years or more of work in a high probability of exposure job. Similar results were seen in additional analyses, controlling for age, study area, employment in electronics occupations and industries, and exposure to carbon tetrachloride, tetrachloroethylene, and trichloroethylene. There was also evidence of an association between astrocytic brain cancer risk and dichloromethane exposure, based on the average intensity score, with an OR = 1.1 (95% CI 0.7, 1.7) for the low-medium intensity group and an OR = 2.2 (95% CI 1.1, 4.1) for the high intensity group, and trend *p*-value <0.05. The combination of high intensity and high duration (21 or more years) was strongly

associated with risk (OR = 6.1 [95% CI 1.5, 28.3]), and a weaker association (OR = 1.4 [95% CI 0.6, 3.2]) was seen for high intensity and shorter duration (2–20 years). The association between cumulative exposure score (low, medium, and high) and astrocytic brain cancer risk was nonlinear (ORs of 0.9, 1.9, and 1.2 in the low, medium, and high exposure categories, respectively).

The strengths of this case-control study include a large sample size, detailed work histories (including information not just about usual or most recent industry and occupation but also about tasks and products for all jobs held since age 15), and comprehensive exposure assessment and analysis along several different dimensions of exposure. The major limitations were the lack of direct exposure information and potential inaccuracy of the descriptions of work histories that were obtained from next-of-kin interviews. Heineman et al. (1994) acknowledge these limitations in the report, and, in response to a letter by Norman and Boggs (1996) criticizing the methodology and interpretation of the study, Heineman et al. (1996) noted that, while the lack of direct exposure information must be interpreted cautiously, it does not invalidate the results. Differential recall bias between cases and controls was unlikely because work histories came from next-of-kin for both groups, the industrial hygienists made their judgments blinded to disease status, and the strong associations that were seen with the exposure measures for dichloromethane were not seen with the other solvents included in the analysis. The relatively strong and statistically significant associations between dichloromethane and astrocytic brain tumors were seen along multiple measures of exposure, suggesting that the results were unlikely to be spurious. Nondifferential misclassification would, on average, attenuate true associations and would be unlikely to result in the types of exposure-response relationships that were observed in this study.

Norman and Boggs (1996) described an apparent inconsistency in the estimated trends in dichloromethane and carbon tetrachloride exposure based on the methodology used in this case-control study (described in more detail in Gomez et al. [1994]). In response, Gomez (1996) noted that the apparent inconsistency was actually due to an error in the labeling of the lines on one of the figures in the report rather than an inconsistency with the estimated trends. Another point raised by Norman and Boggs (1996) was that the Heineman et al. (1994) findings were surprising in light of the lack of brain carcinogenesis in animals. In response, Heineman et al. (1996) pointed out that carcinogens commonly cause different cancers in animals and humans. It can also be noted that brain tumors are exceedingly rare in animal bioassays (Sills et al., 1999). Norman and Boggs (1996) also suggested that the results of the Heineman et al. (1994) study be given no weight when compared with the results of the cohort studies. The authors responded by pointing out that the cohort studies had low statistical power and large CIs around their point estimates but were not inconsistent with an association between dichloromethane and brain cancer (Heineman et al., 1996). This point is strengthened further by the more recent results from the Rochester, New York, Eastman Kodak cohort (Hearne and Pifer, 1999), described

previously, since an increased SMR for brain and CNS cancers was seen in the longer follow-up period of this cohort.

In another case-control study of brain cancer and dichloromethane exposure, Cocco et al. (1999) identified 12,980 female cases of cancer of the brain and CNS through the underlying cause of death listing (ICD codes 191 and 192) on death certificates from 24 states from 1984 to 1992. (This collection of death certificates is a data set created by the National Center for Health Statistics, NIOSH, and the National Cancer Institute to facilitate research on occupational exposures and mortality risk.) The cases included 161 women with meningioma (ICD-9 codes 192.1, 192.3). Four women who died of nonmalignant diseases, excluding neurological disorders, were chosen as controls for each case. The controls were frequency matched to the cases by state, race, and 5-year age group. Occupation data were based on the occupation fields in the death certificates. This job was coded based on the three-digit industry and three-digit occupation (Department of Census) codes. The investigators developed job exposure matrices that were applied to these industry/occupation codes. The job exposure matrices included probability and intensity scores for 11 occupational hazards, one of which was dichloromethane, but also included other solvents, electromagnetic fields, chlorinated aliphatic hydrocarbons, benzene, lead, nitrosamines, insecticides, herbicides, and public contact. The investigators used logistic regression models to estimate ORs, adjusting for each workplace exposure, marital status, three levels of socioeconomic status (based on occupation), and age at death. For each chemical, four levels of intensity and probability were defined (unexposed, low, medium, and high).

A weak association between dichloromethane exposure and brain/CNS cancer was seen (OR 1.2 [95% CI 1.1, 1.3]) (Cocco et al., 1999). There was no exposure-related trend in the association between probability or intensity of exposure and brain cancer. A similar but more imprecise association was seen with meningioma cancer (OR 1.2 [95% CI 0.7, 2.2]). There were too few cases of meningioma to stratify by exposure probability and intensity.

The major limitations of this study are the use of mortality rather than incidence data and the reliance on occupation data from death certificates. The death certificate occupation data are based on "usual" occupation, which may be more prone to misclassification in studies of women because of gender-related differences in work patterns (i.e., shorter duration jobs for women compared with men). A relatively broad job exposure matrix was applied to the job information, and typically more generic job exposure matrices result in less sensitive assessment with limited ability to detect exposure-response trends (Teschke et al., 2002). Nondifferential misclassification of outcome and exposure would generally result in attenuated effect estimates.

4.1.3.6.2. ***Case-control studies of breast cancer.*** Cantor et al. (1995) conducted a case-control study of occupational exposures and breast cancer, using the 24 state (1984–1989) death certificate data described in the previous section. Cases were women with breast cancer coded as

the underlying cause of death (ICD-9 code 174). Four female controls per case were selected from all noncancer deaths, frequency matched by age (5-year age groups) and ethnicity (black, white). The occupation listed on the death certificate was coded based on the three-digit industry and three-digit occupation (Department of Census) codes, and this was used with a job exposure matrix developed by the investigators to assess 31 workplace exposures, one of which was dichloromethane. Four exposure probability and three exposure level scores were assigned. ORs for probability and level were calculated for each ethnic group, adjusting for age at death and a measure of socioeconomic status (based on occupation). After excluding subjects whose death certificate occupations were listed as homemaker, there were 29,397 white cases and 4,112 black cases (total 33,509) and 102,955 white controls and 14,839 black controls (total 117,794).

There was little evidence of an association between exposure probability and breast cancer mortality using the probability exposure metric. The ORs were 1.05 (95% CI 0.97, 1.1) and 0.76 (95% CI 0.3, 2.0) in probability level 3 and level 4, respectively, for white women and 1.13 (95% CI 0.9, 1.4) in probability level 3 for black women. (There were too few black women in exposure probability level 4 for analysis.) Weak associations were seen with exposure level. In white women, an OR of 1.17 (95% CI 1.1, 1.3) was seen with the highest exposure level, and in black women the OR in this exposure group was 1.46 (95% CI 1.2, 1.7). In the analysis that jointly considered exposure level and probability ratings but excluded the lowest probability of exposure, the OR for the highest category of exposure level was 1.28 in whites ($p < 0.05$) and 1.21 in blacks.

As with the Cocco et al. (1999) case-control study that used a similar methodology, the limitations of this study include the use of an outcome defined by mortality rather than incidence, use of usual occupation information as recorded in death certificates, and use of a very broad job exposure matrix to classify 31 different exposures. Although information on pregnancy and lactation history (known risk factors for breast cancer) was not available, the authors did adjust for socioeconomic status by using the occupation data, which may have corrected for some of the potential confounding due to reproductive history.

4.1.3.6.3. ***Case-control studies of pancreatic cancer.*** Kernan et al. (1999) conducted a case-control study of 63,097 pancreatic cancer cases, using the 24-state (1984–1993) death certificate data. The diagnosis of pancreatic cancer was based on underlying cause of death (ICD-9 code 157). Four controls who had died during the same time period of causes other than cancer were selected for each case, frequency-matched by state, race, gender, and 5-year age group (n = 252,386). Usual occupation and industry, based on the occupation data in the death certificate, were coded by using the three-digit (Department of Census) codes. A job-exposure matrix was used with the industry and occupation codes to evaluate exposure intensity and probability (each categorized as high, medium, or low) for formaldehyde, dichloromethane,

10 other solvents, and a combined "organic solvents" measure. Race- and gender- specific analyses were conducted by using logistic regression to estimate ORs and 95% CIs, adjusting for age, marital status (ever, never married), residential area (metropolitan, nonmetropolitan), and region (east, south central, south, and west).

The point estimates for the ORs in the low, medium, and high intensity categories in the four race-gender groups ranged from 0.8 to 1.3, with no exposure-effect trend seen in any group. The only statistically significant OR was for high exposure intensity in white females (OR 1.3 [95% CI 1.1–1.6]), with ORs of 1.0 (95% CI 0.9, 1.1) for medium intensity and 1.1 (1.0, 1.2) for low intensity in this group. An elevated OR was seen with high exposure probability in black males (OR 2.2 [95% CI 1.0, 4.8]) but not in white females (OR 1.0 [95% CI 0.8, 1.4]) or white males (OR 1.0 [05% CI 0.8, 1.3]), and the ORs were 0.9 for medium exposure probability in these three groups. There were relatively few black females in this study, resulting in imprecise estimates (OR 2.0 [95% CI 0.8, 5.4] for medium exposure and OR 1.5 [95% CI 0.6, 3.6] for high exposure).

The limitations of this study, as with the other case-control studies that used the 24-state death certificate data set, include the reliance on cause of death data from death certificates rather than medical-record validated incidence data and the use of death certificate occupation data. The job exposure matrix used with the occupation data was more focused than those used in Cocco et al. (1999) and Cantor et al. (1995). Although the analysis adjusted for some sociodemographic characteristics, it did not include measures of smoking history or diabetes, which are known risk factors for pancreatic cancer (Lowenfels and Maisonneuve, 2005).

4.1.3.6.4. ***Case-control studies of renal cancer.*** Dosemeci et al. (1999) reported data from a population-based case-control study of the association between occupation exposures and renal cancer risk. The investigators identified newly diagnosed patients with histologically confirmed renal cell carcinoma from the Minnesota Cancer Surveillance System from July 1, 1988, to December 31, 1990. The study was limited to white cases, and age and gender-stratified controls were ascertained by using random digit dialing (for subjects ages 20–64) and from Medicare records (for subjects 65–85 years). Of the 796 cases and 796 controls initially identified, 438 cases (273 men, 165 women) and 687 controls (462 men, 225 women) with complete personal interviews were included in the occupational analysis.

Data were obtained through in-person interviews that included demographic variables, residential history, diet, smoking habits, medical history, and drug use. The occupational history included information about the most recent and usual industry and occupation (coded using the standard industrial and occupation codes, Department of Commerce), job activities, hire and termination dates, and full- and part-time status. A job exposure matrix developed by the National Cancer Institute was used with the coded job data to estimate exposure status to dichloromethane and eight other chlorinated aliphatic hydrocarbons.

ORs were adjusted for age, smoking, hypertension and use of drugs for hypertension, and body mass index. No association between renal cell carcinoma and exposure to dichloromethane was observed in men (OR 0.85 [95% CI 0.6, 1.2]), women (OR 0.95 [95% CI 0.4, 2.2]), or both sexes combined (OR 0.87 [95% CI 0.6, 1.2]).

A strength of this study includes the use of incident cases of renal cancer from a defined population area, with confirmation of the diagnosis using histology reports. The occupation history was based on usual and most recent job, in combination with a relatively focused job exposure matrix. In contrast to the type of exposure assessment that can be conducted in cohort studies within a specific workplace, however, exposure measurements, based on personal or workplace measurements, were not used, and a full lifetime job history was not obtained.

4.1.3.6.5. ***Case-control studies of rectal cancer.*** Dumas et al. (2000) reported data from a case-control study of occupational exposures and rectal cancer conducted in Montreal, Quebec, Canada. The investigators identified 304 newly diagnosed cases of primary rectal cancer, confirmed on the basis of histology reports, between 1979 and 1985; 257 of these participated in the study interview. One control group (n = 1,295) consisted of patients with other forms of cancer (excluding lung cancer and other intestinal cancers) recruited through the same study procedures and time period as the rectal cancer cases. A population-based control group (n = 533), frequency matched by age strata, was drawn by using electoral lists and random digit dialing. The occupational assessment consisted of a detailed description of each job held during the working lifetime, including the company, products, nature of work at site, job activities, and any additional information from the interviews that could furnish clues about exposure. The percentage of proxy respondents was 15.2% for cases, 19.7% for other cancer controls, and 12.6% for the population controls.

A team of industrial hygienists and chemists blinded to subjects' disease status translated jobs into potential exposure to 294 substances with three dimensions (degree of confidence that exposure occurred, frequency of exposure, and concentration of exposure). Each of these exposure dimensions was categorized into none, any, or substantial exposure. Logistic regression models adjusted for age, education, proxy versus subject responder status, cigarette smoking, beer drinking, and body mass index. Using the cancer control group, the OR for any exposure to dichloromethane was 1.2 (95% CI 0.5, 2.8) and the OR for substantial exposure (confident that exposure occurred with 5 or more years of exposure at medium or high frequency and concentration) was 3.8 (95% CI 1.1, 12.2). The results using the population-based control group for this exposure were not presented.

The strengths of this study were the large number of incident cases, specific information about job duties for all jobs held, and a definitive diagnosis of rectal cancer. However, the use of the general population (rather than a known cohort of exposed workers) reduced the likelihood that subjects were exposed to dichloromethane, resulting in relatively low statistical power for

the analysis. The job exposure matrix, applied to the job information, was very broad since it was used to evaluate 294 chemicals.

4.1.3.6.6. ***Case-control studies of childhood leukemia.*** Infante-Rivard et al. (2005) examined the association between maternal occupational exposures, before and during pregnancy, and risk of childhood acute lymphoblastic leukemia (ICD-9 code 204.0) by using data from a population-based case-control study in Quebec, Canada. Incident cases diagnosed from 1980–2000 were identified from the cancer hospitals in the province, and diagnosis was confirmed based on clinical records from an oncologist or hematologist. Between 1980 and 1993, cases ages 0–9 years at diagnosis were included, and from 1994 to 2000 the age range was expanded to 14 years. The number of eligible cases identified was 848, and, of these, 790 parents (93%) participated in the study. Population-based controls, individually matched to the sex and age at diagnosis of the cases, were identified from government registries of all children in the province (1980–1993) and the universal health insurance files (1994–2000). The parents of 790 (86%) of the 916 eligible controls who were identified participated in the study.

Data were collected by using a structured telephone interview. Some information (i.e., job title, dates, type of industry, industry name and address) was obtained for all jobs held since age 18, and additional information (e.g., materials and machines used, typical activities) was obtained for jobs held by the mother from 2 years before the pregnancy through the birth of the child. Specialized exposure modules were also used to collect information about specific jobs (e.g., nurse, waitress, hair dresser, textile dry cleaner). All of this information was reviewed by chemists and industrial hygienists, blinded to case-control status, to classify exposure to over 300 chemicals, although the primary focus of the study was on solvents (21 individual substances, including dichloromethane, and six mixtures). The exposure assessment included ratings of confidence (possible, probable, and definite), frequency of exposure during a normal workweek (<5, 5–30, or >30% of the time), and level of concentration (low = slightly above background, high = highest possible exposure in the study population, and medium for in-between levels).

A weak association was seen between any dichloromethane exposure during the 2 years before pregnancy up to the birth and risk of leukemia in the child (OR 1.34 [95% CI 0.54, 3.34]), and results were similar when limited to exposures during pregnancy. Stronger associations were seen with probable or definite exposure (OR 3.22 [95% CI 0.88, 11.7]) compared with possible or no exposure. The estimates for categories based on concentration and frequency were similar but there was no evidence for an increasing risk with increasing exposure level.

4.1.3.7. ***Summary of Cancer Studies by Type of Cancer***

The cohort and case-control studies with data relevant to the issue of dichloromethane exposure and cancer risk are summarized in Tables 4-9 and 4-10, respectively. The strongest of the cohort studies, in terms of design, are two of the triacetate film base production cohorts

(Cohort 1 in New York and the United Kingdom cohort, reported in Hearne and Pifer [1999] and Tomenson et al. [1997], respectively). These are the cohorts with the most extensive exposure assessment information. The start of eligibility for cohort entrance corresponds with the beginning of the time when the exposure potential at the work site began, and the follow-up period is relatively long (mean >25 years). Although Cohort 2 of the New York film base production study has similar exposure data and follow-up, this cohort was limited to workers employed between 1964 and 1970 and therefore would have missed anyone leaving (possibly because of illness or death) before this time. In addition, because of the overlap between Cohort 1 and Cohort 2, including both cohorts in an evaluation would be double counting experiences of some individuals. Several limitations of the triacetate film base production cohorts should be noted, however. One of these limitations concerns the generalizability of the results, given the relatively low exposure level (mean 8-hour TWA <40 ppm) compared to the other cellulose triacetate fiber production cohorts (Gibbs et al., 1996; Ott et al., 1983a). Exposures in small, poorly ventilated work areas are also often much higher than those seen in these film base production cohorts (Estill and Spencer, 1996; Anundi et al., 1993). Other limitations include the limited power to detect a risk of low-incidence cancers (including brain and leukemia), the lack of women and thus lack of data pertaining to breast cancer, and the use of mortality rather than incidence data. Although the exposure levels in the cohorts involved in cellulose triacetate fiber production were much higher than those of the film production cohorts, the duration of exposure was relatively short in the South Carolina cohort (Lanes et al., 1993), and the majority of workers were missing job history data. In the Maryland triacetate fiber production plant, duration of exposure was not reported and the length of follow-up was relatively short (mean, 17 years) (Gibbs et al., 1996). Also, the cohort began in 1970, even though production began in 1955, and the missing personnel records made it impossible to recreate an inception cohort. The exposure assessment in the study of civilian Air Force base workers (Blair et al., 1998) allowed for only a dichotomized classification of exposure, and there was considerable exposure to other solvents among these workers. This Air Force base study was the largest of the cohort studies that included women and presented data pertaining to breast cancer.

Table 4-9. Summary of cohort studies of cancer risk and dichloromethane exposure

Cohort	Total n, exposure level[a] and duration, length of follow-up	Inclusion criteria[b]	Exposure assessment; Outcome assessment	Results[c]
Hearne and Pifer (1999) Cellulose triacetate film base production; New York Cohort 1	n = 1,311 men Mean, 39 ppm mean duration, 17 yr mean follow-up, 35 yr	Began working after 1945; worked at least 1 yr	Work history (job records) and personal/air monitoring; death certificate (underlying causes)	Elevated mortality risks seen for brain cancer, Hodgkin's disease, and leukemia (SMRs around 2.0); no risk for liver, lung, or pancreatic cancer (see Table 4-4)
Cohort 2	n = 1,013 men mean, 26 ppm mean duration, 24 yr mean follow-up, 26 yr	Employed at least 1 yr between 1964 and 1970 (potential exposure began 1946)	Work history (job records) and personal/air monitoring; death certificate (underlying causes)	Elevated mortality risks seen for brain cancer, Hodgkin's disease, leukemia, and pancreatic cancer (SMRs between 1.5 and 3); no risk for liver or lung cancer (see Table 4-4)
Tomenson et al. (1997) Cellulose triacetate film base production; United Kingdom	n = 1,473 men mean, 19 ppm mean duration, 9 yr mean follow-up, 27 yr	Employed anytime between 1946 and 1988	Work history (job records) and personal/air monitoring; death certificate (underlying causes)	Elevated mortality risks seen for brain cancer (SMR 1.45); weak elevation for leukemia; no risk for liver, lung, or pancreatic cancer (see Table 4-5)
Lanes et al. (1993) Cellulose triacetate fiber production; South Carolina	n = 551 men and 720 women (total n = 1,271); median 140, 280, and 475 ppm in low, moderate, and high, respectively; 56% <5 yr work duration; mean follow-up, ~28 yr	Worked at least 3 months in the preparation or extrusion areas from 1954 to 1977	Job history data and personal/air monitoring of specific areas (but job history data available for 37%); death certificate (underlying and contributing causes)	Elevated mortality risk for liver cancer (SMR 2.98, lower than seen in earlier study of this cohort); no risk for lung, pancreatic, or brain cancer (see Table 4-7)
Gibbs et al. (1996) Cellulose triacetate fiber production; Maryland	n = 1,931 men and 978 women (total n = 2,909); 50–100 ppm in low and 350–700 ppm in high exposure; duration not reported; mean follow-up 17 yr	Employed on or after January 1, 1970, for at least 3 months (potential exposure began 1955)	Work history (job records) and personal/air monitoring; death certificate (fields used not stated)	Elevated mortality risk for prostate cancer (men, SMRs 1.4 and 1.8)), cervical cancer (women, SMR ≥ 3.0), and lung cancer in women (high exposure, SMR 2.3), but not in men; weak risk for liver cancer, no risk for pancreatic or brain cancer (see Table 4-8)

Table 4-9. Summary of cohort studies of cancer risk and dichloromethane exposure

Cohort	Total n, exposure level[a] and duration, length of follow-up	Inclusion criteria[b]	Exposure assessment; Outcome assessment	Results[c]
Blair et al. (1998) Air Force Base, Utah	n = 10,461 men and 3,605 women (total n = 14,066)[d] dichotomized (yes, no) exposure duration not reported mean follow-up ~29 yr	Employed at least 1 yr from 1952 to 1956 (potential exposure began 1939)	Work history (job records) and industrial hygiene assessment based on work site review (dichotomized exposure); (underlying and contributing causes)	Elevated mortality risk for non-Hodgkin's lymphoma (RR 3.0) and multiple myeloma (RR 3.4) in men, and breast cancer in women (RR 3.0) (see section 4.1.3.5)

[a]8-hour TWA.
[b]If dichloromethane was used at the plant before the first date of entrance into the cohort, the year that potential exposure began is noted.
[c]Results are described as elevated if SMR was around 1.5 or higher. There is limited statistical power for these cause-specific analyses in these cohort studies; the statistical significance of individual estimates is not presented in this table. RR = relative risk.
[d]Includes whites and unknown race.

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Table 4-10. Summary of case-control studies of cancer risk and dichloromethane exposure

Cancer type, reference	Location n cases, n controls (source), demographic group	Exposure assessment	Results
Brain Heineman et al. (1994)	Louisiana, New Jersey, Philadelphia 300 cases, 320 controls (death certificates); cancer confirmed by hospital records; white men	Job exposure matrix applied to detailed information on all jobs held (at least 1 year) since age 15, as obtained from next-of-kin interviews; probability, duration, intensity, and cumulative exposure scores; six solvents evaluated	OR 1.3 for any exposure; increased risk with increased probability (trend *p*-value <0.05, OR 2.4 for high probability), increased duration, increased intensity; strongest effects seen in high probability plus high duration (OR 6.1) or high intensity and high duration (OR 6.1) combinations; no association with cumulative exposure score (see section 4.1.3.6.1)
Brain Cocco et al. (1999)	24 states, U.S. 12,980 cases, 51,920 controls (death certificates); women	Job exposure matrix applied to death certificate occupation; probability, and intensity scores; 11 exposures evaluated	Weak association overall (OR 1.2), no trend with probability or intensity scores (see section 4.1.3.6.1)
Breast Cantor et al. (1995)	24 states, U.S. 33,509 cases, 117,794 controls (death certificates); black and white women	Job exposure matrix applied to death certificate job data, probability, and exposure level; 31 substances evaluated	Little evidence of association with exposure probability; weak association with exposure level in whites and in blacks (see section 4.1.3.6.2)
Pancreas Kernan et al. (1999)	24 states, U.S. 63,037 cases, 252,386 controls (death certificates); black and white men and women	Job exposure matrix applied to death certificate occupation, probability, and intensity scores; 11 chlorinated solvents and formaldehyde evaluated	Little evidence of associations with intensity or probability (see section 4.1.3.6.3)
Kidney Dosemeci et al. (1999)	Minnesota 438 incident cases (Minnesota cancer registry), 687 controls (random digit dialing and Medicare records); cancer confirmed by histology; men and women	Job exposure matrices applied to most recent and usual job, as ascertained from interviews; nine solvents evaluated	No evidence of increased risk associated with dichloromethane (OR 0.85 in men, 0.95 in women) (see section 4.1.3.6.4)
Rectum Dumas et al. (2000)	Montreal, Canada 257 incident cases, 1,295 other cancer controls from 19 hospitals; 533 population-based controls (electoral rolls and random digit dialing), cancer confirmed by histology; men	Job exposure matrix applied to detailed information on all jobs held, as ascertained from interviews; 294 substances evaluated	Little evidence of an association with any exposure (OR 1.2), but increased risk in a small, "substantial exposure" group (OR 3.8) (using cancer controls; analysis of population controls not given for this exposure) (see section 4.1.3.6.5)

Table 4-10. Summary of case-control studies of cancer risk and dichloromethane exposure

Cancer type, reference	Location n cases, n controls (source), demographic group	Exposure assessment	Results
Childhood leukemia (acute lymphoblastic leukemia) Infante-Rivard et al. (2005)	Quebec, Canada 790 incident cases (hospitals—all provinces), 790 population-based controls (government population registries); cancer based on oncologist or hematologist diagnosis ages 0–14,[a] both sexes	Systematic review of detailed information on all jobs held by the mother from 2 years before pregnancy through birth of the child; 21 individual substances and six mixtures evaluated (mostly solvents); confidence, frequency, and concentration of exposure rated	Little evidence of association with any exposure (OR 1.3), but stronger associations (OR > 3.0, referent group = possible/no exposure) with probable or definite and with combinations of frequency and concentration (see section 4.1.3.6.6)

[a]From 1980 to 1993, study was limited to diagnoses of ages 0–9, but this was expanded between 1994 and 2000 to ages 0–14.

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Case-control studies offer the potential for increased statistical power for assessing associations with relatively rare cancers, such as brain cancer and leukemia. Case-control studies are often designed to examine incidence rather than mortality, which is of particular importance in etiologic research for diseases with relatively high survival rates and diseases in which survival may be strongly related to factors that are difficult to adjust for without detailed data collection (e.g., access to health care). There is a considerable range in the detail and quality of the exposure assessment used in case-control studies, however. Case-control studies rarely include specific measurements taken at specific work sites of individual study participants. Although it is more difficult to determine absolute exposure levels without these individual measurements, the exposure assessment methodology used in case-control studies can result in useful between-group comparisons of risk if the intra-group variability is less than the inter-group variability in potential exposure levels. Among the case-control studies with data pertaining to cancer risk and dichloromethane exposure, the two studies with the strongest designs are the study of brain cancer by Heineman et al. (1994) and the study of childhood leukemia by Infante-Rivard et al. (2005). These are the studies that obtained detailed information about all jobs held (rather than just the usual or most recent job), focused on a relatively small number of exposures, and used medical record data to confirm the diagnosis. Heineman et al. (1994) obtained the work history from interviews with next-of-kin, however, which is most likely to have resulted in nondifferential misclassification of exposure, and thus attenuation in the observed associations. The use of death certificate data to classify disease and occupational exposures in the three studies using the large 24 state death certificate database (brain cancer: Cocco et al. [1999]; breast cancer: Cantor et al. [1995]; pancreatic cancer: Kernan et al. [1999]) is also likely to have resulted in nondifferential misclassification of both outcome and exposure (and thus attenuated associations).

Considering the issues described above with respect to the strengths and limitations of the available epidemiologic studies, a summary of the epidemiologic evidence relating to dichloromethane exposure and specific types of cancer can be made, as described below. The available epidemiologic data suggest an association between dichloromethane and brain cancer and liver cancer, but not lung cancer.

4.1.3.7.1. ***Brain and CNS cancer.*** An increased risk of brain and CNS cancers was seen in the strongest cohort studies; SMRs were 2.16 in Cohort 1 in New York (Hearne and Pifer, 1999) and 1.45 in the United Kingdom cohort (Tomenson et al., 1997). These estimates are based on a small number of observations (six cases in New York and four in the United Kingdom) and so are relatively imprecise. It is only in the latest follow-up of the New York film base production cohort that an elevated SMR was observed, further suggesting that the statistical power of the other cohort studies for examining risk of this disease may be quite low. Two case-control studies of dichloromethane exposure and brain cancer have been conducted (Cocco et al., 1999;

Heineman et al., 1994). The Heineman et al. (1994) study, which is the stronger study in terms of exposure assessment strategy and confirmation of diagnosis, reported relatively strong trends with increasing probability, duration, and intensity measures of exposure, but a nonlinear trend was seen with the cumulative exposure metric. This difference could reflect a more valid measure of relevant exposures in the brain from the intensity measure, as suggested by the study in rats reported by Savolainen et al. (1981) in which dichloromethane levels in the brain were much higher with a higher intensity exposure scenario compared with a constant exposure period with an equivalent TWA (see section 3.2). The available epidemiologic studies provide some evidence of an association between dichloromethane and brain cancer, and this area of research represents a data gap in the understanding of the carcinogenic potential of dichloromethane.

4.1.3.7.2. ***Liver and biliary duct cancer.*** Liver and biliary duct cancer are relatively uncommon (age-adjusted incidence 6.2 per 100,000 person-years) (SEER website, seer.cancer.gov, accessed April 2006), so it is difficult to study in most occupational cohorts of limited size. The cohort study with the higher exposures, the Rock Hill, South Carolina, triacetate fiber production plant, suggested an increased risk of liver cancer (Lanes et al., 1993, 1990). The SMR for liver and bile duct cancer was 2.98 (95% CI 0.81, 7.63) in the latest update of this cohort. This observation was based on four cases; two of these cases were biliary duct cancers. As the follow-up period has increased, the strength of this association has decreased, although it is relatively strong (albeit with wide CIs). The decrease in the SMR with increasing follow-up reflects the increase in number of expected cases, because the four observed cases were seen earlier in the follow-up period. No other cohort study has reported an increased risk of liver cancer mortality, although it should be noted that there is no other inception cohort study of a population with exposure levels similar to those of the Rock Hill plant, and no data from a case-control study of liver cancer are available pertaining to dichloromethane exposure. The available epidemiologic studies, with biological plausibility inferred from the results from studies in mice and female rats (see section 4.2) (NTP, 1986; Serota et al., 1986a, b; Nitschke 1988a), provide some evidence of an association between dichloromethane and liver and biliary duct cancer, although it should be noted that this evidence is based on very limited epidemiologic data.

4.1.3.7.3. ***Lung cancer.*** In the stronger cohort studies (Cohort 1 in the New York Eastman Kodak Company triacetate film production study reported by Hearne and Pifer [1999] and the United Kingdom triacetate film production study reported by Tomenson et al. [1997]), the SMRs for lung cancer were well below 1.0. The New York study had also obtained data on smoking history that indicated it was unlikely that differences in smoking could be masking an effect of dichloromethane (Hearne et al., 1987). Lung cancer is a common cancer (age-adjusted incidence 61 per 100,000 person-years) (SEER website, seer.cancer.gov, accessed April 2006), so the expected rates, even in small cohorts, are based on relatively robust estimates. The only group in

any study that had an increased risk for lung cancer was the high-exposure women in the triacetate fiber production cohort in Maryland (Gibbs et al., 1996). However, this was based on only two cases and was a highly imprecise estimate (SMR 2.3 [95% CI 0.28, 8.3]). No case-control study of dichloromethane exposure and lung cancer risk is available. The available epidemiologic studies do not provide evidence for an association between dichloromethane and lung cancer, although it should be noted that the studies with the best designs are limited to relatively low exposure levels.

4.1.3.7.4. ***Pancreatic cancer.*** An early study (Hearne et al., 1990) of Cohort 2 of the New York triacetate film production cohort had reported 8 observed and 4.2 expected pancreatic cancer deaths, for a twofold increased SMR ($p = 0.13$). This association was reduced in the subsequent follow-up (SMR 1.5 [95% CI 0.7, 3.0]) (Hearne and Pifer, 1999) but was not seen in the more methodologically sound Cohort 1 (SMR 0.92) or in any of the other cohorts. A meta-analysis of the cohort studies (using the data of Hearne et al. [1990]) reported a summary association of 1.42 (95% CI 0.80, 2.53) (Ojajärvi et al., 2001). This summary measure would be further reduced with the updated data for Cohort 2 and the addition of Cohort 1 from Hearne and Pifer (1999). The only case-control study of pancreatic cancer mortality risk and dichloromethane exposure (based on death certificate data) did not report consistent patterns with respect to intensity or exposure among the race-sex groups studied. The available epidemiologic studies do not provide evidence for an association between dichloromethane and pancreatic cancer.

4.1.3.7.5. ***Leukemia and lymphoma.*** Each of the individual hematopoietic cancers is relatively uncommon, with age-adjusted incidence rates of 5 per 100,000 person-years or less (SEER website, seer.cancer.gov, accessed April 2006). The relatively inconsistent (point estimates ranging from 0.50 or less to 2.0 or higher) and imprecise measures of association between dichloromethane exposure and non-Hodgkin's lymphoma, Hodgkin's lymphoma, myeloma, and leukemia are thus expected, given the relatively small size of the available cohort studies. Only one case-control study of any of these diseases and dichloromethane is available, and this is a study of childhood leukemia (acute lymphoblastic leukemia) in relation to maternal occupational history (Infante-Rivard et al., 2005). This is a large, population-based study of confirmed incident cases of leukemia, with a detailed exposure assessment pertaining to the period before and during pregnancy. A threefold increased risk was seen with probable or definite exposure (OR 3.22 [95% CI 0.88, 11.7]) compared with possible or no exposure. The available epidemiologic studies do not provide an adequate basis for the evaluation of the role of dichloromethane in any of the specific hematopoietic cancers because of the small size of the cohort studies and the relative lack of case-control studies pertaining to these outcomes.

4.1.3.7.6. ***Breast cancer.*** Only one large cohort study included women and reported data pertaining to breast cancer risk (Blair et al., 1998), and this is a cohort with a limited exposure assessment (dichotomized) and multiple exposures. A relatively strong association was seen between dichloromethane exposure and breast cancer mortality in this study (rate ratio 3.0 [95% CI 1.0, 8.8]). Similar associations were seen with several other chemicals, and the potential effect of confounding and misclassification of these exposures may have biased the estimate in either direction. The only case-control study of breast cancer risk and dichloromethane exposure used the 24-state death certificate data to classify exposure and disease. The available epidemiologic studies do not provide an adequate basis for the evaluation of the role of dichloromethane in breast cancer because there are currently no cohort studies with adequate statistical power and no case-control studies with adequate exposure methodology to examine this relationship.

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

4.2.1. Oral Exposure: Overview of Noncancer and Cancer Effects

Results from studies of animals exposed by the oral route for short-term, subchronic, and chronic durations identify the liver and the nervous system as the most sensitive targets for noncancer toxicity from repeated oral exposure to dichloromethane. In a 90-day exposure study, nonneoplastic histopathologic changes in the liver were observed in F344 rats exposed to drinking water doses of ≥166 mg/kg-day (males) or ≥209 mg/kg-day (females) (Kirschman et al., 1986). Similar changes were seen in F344 rats in a 2-year exposure of ≥50 mg/kg-day (Serota et al., 1986a).

The 2-year oral exposure study in F344 rats did not produce evidence of increasing incidence of liver tumors across all of the dose groups in males or females (Serota et al., 1986a). In females, however, a jagged stepped pattern of increasing incidence was observed. In a parallel study in $B6C3F_1$ mice (Serota et al., 1986b; Hazelton Laboratories, 1983), a clearer trend with respect to hepatic cancer was seen in males but not females.

None of the chronic oral exposure studies included a systematic measurement of potential neurological effects. One 14-day study focusing on neurobehavioral changes is available, however. Changes in autonomic, neuromuscular, and sensorimotor functions were observed in F344 rats exposed for 14 days to gavage doses ≥337 mg/kg-day (Moser et al., 1995) (see section 4.4.3 for more details).

No effects on reproductive parameters were observed in Charles River CD rats exposed for 90 days to gavage doses as high as 225 mg/kg-day (General Electric Co., 1976) or in pregnant F344 rats exposed to gavage doses of up to 450 mg/kg-day on gestation days (GDs) 6–19 (Narotsky and Kavlock, 1995). However, no oral exposure studies examining developmental neurobehavioral effects have been conducted (see section 4.3 for more details).

4.2.1.1. *Toxicity Studies of Subchronic Oral Exposures: Hepatic Effects*

Kirschman et al. (1986) examined the toxicity of dichloromethane in a 90-day drinking water study in F344 rats (20/sex/dose level). The nominal concentration of dichloromethane in the water was 0.15, 0.45, or 1.5%. Based on BW and water consumption data, average intakes were reported to be 0, 166, 420, or 1,200 mg/kg-day for males and 0, 209, 607, or 1,469 mg/kg-day for females. Clinical chemistry tests (hematological and chemical variables in samples of blood and urine) and tissue histopathology were evaluated in groups of five rats/sex/dose level after 1 month of treatment. These endpoints were also evaluated in the remaining rats sacrificed after 90 days of exposure.

Exposure to dichloromethane did not affect mortality or cause adverse clinical signs of toxicity. Gross necropsy was also unremarkable. Reported changes in mean values for clinical chemistry variables, compared with controls, included elevated serum ALT activities for all treated males at 1 month and for the high-dose females at 3 months, elevated serum AST activity in high-dose females at 3 months, elevated serum lactate dehydrogenase activities in mid- and high-dose females at 3 months, and decreases in serum concentrations of fasting glucose, cholesterol, and triglycerides in all exposed groups of both sexes at 1 and 3 months. Actual values for clinical chemistry variables, however, were not presented in the report.

No histopathologic alterations were seen in tissues after 1 month of treatment (a detailed description of tissues examined was not presented). In rats exposed for 3 months, exposure-related histopathologic changes were restricted to the liver. Elevated, statistically significant, incidences of hepatocytic vacuolation were observed in all exposed male groups and in the mid- and high-dose female groups (see Table 4-11). The most frequently observed vacuolation was described as generalized and occurring throughout the lobule, and Oil Red-O-staining indicated most were lipid-containing vacuoles. The incidences of generalized vacuolation scored as mild or moderate were higher in all of the female dose groups compared with the controls. The authors stated that the no-observed-adverse-effect level (NOAEL) based on this study is less than 200 mg/kg-day and the lowest-observed-adverse-effect level (LOAEL) for males was 166 mg/kg-day. The authors did not explicitly provide a LOAEL for females. The results indicate that 166 mg/kg-day and 209 mg/kg-day were the LOAELs for liver effects in male and female rats, respectively.

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Table 4-11. Incidences of histopathologic changes in livers of male and female F344 rats exposed to dichloromethane in drinking water for 90 days

Lesion, by sex	Controls	Low dose	Mid dose	High dose
Males—n per group[a]	15	15	15	15
Estimated mean intake (mg/kg-day)	0	166	420	1,200
Number (%) with				
Hepatocyte vacuolation (generalized, centrilobular, or periportal)	1 (7)	10[b] (67)	9[b] (60)	7[b] (47)
Generalized vacuolation severity:	0 (0)	5[b] (33)	8[b] (53)	6[b] (40)
minimal	0	4	7	6
mild	0	0	1	0
moderate	0	1	0	0
Centrilobular severity:	0 (0)	1 (7)	0 (0)	2 (13)
minimal	0	1	0	0
mild	0	0	0	2
moderate	0	0	0	0
Hepatocyte degeneration	0 (0)	0 (0)	0 (0)	2 (13)
Focal granuloma	1 (7)	0 (0)	0 (0)	1 (7)
Females—n per group[a]	15	15	15	15
Estimated mean intake (mg/kg-day)	0	209	607	1,469
Number (%) with				
Hepatocyte vacuolation (generalized, centrilobular, or periportal)	6 (40)	13[b] (87)	15[b] (100)	15[b] (100)
Generalized vacuolation severity:	5 (33)	13[b] (87)	15[b] (100)	15[b] (100)
minimal	5	8	6	8
mild	0	4	5	6
moderate	0	1	4	1
Centrilobular severity:	0 (0)	0 (0)	1 (7)	11[b] (28)
minimal	0	0	0	2
mild	0	0	1	4
moderate	0	0	0	3
marked	0	0	0	2
Hepatocyte degeneration	0 (0)	0 (0)	0 (0)	12[b] (80)
Focal granuloma	0 (0)	0 (0)	4 (27)[c]	6[b] (40)

[a]20 per group; 5 sacrificed at 1 month; these endpoints for the remaining 15 per group.
[b]Statistical significance testing not reported by authors; Fisher's exact test for comparison with control *p*-- value <0.05 (two-sided).
[c]Statistical significance testing not reported by authors; Fisher's exact test for comparison with control *p* - value <0.10 (two-sided). Authors stated LOAEL = 166 mg/kg-day in males but did not explicitly provide LOAEL for females; NOAEL is less than 200 mg/kg-day.

Source: Kirschman et al. (1986).

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3524 Kirschman et al. (1986) conducted a similar 90-day study in B6C3F$_1$ mice (20/sex/dose
3525 level). The estimated average intakes were 0, 226, 587, or 1,911 mg/kg-day for males and 231,
3526 586, or 2,030 mg/kg-day for females. Six mice (two controls, two low dose, and two mid dose)
3527 died during the study from unknown causes. Administration of dichloromethane did not cause
3528 adverse clinical signs of toxicity or affect food consumption, ophthalmology, or serum ALT
3529 activity. Gross necropsy examinations also were unremarkable.
3530 Histopathologic evaluation of tissues from mice killed after 1 month of treatment did not
3531 reveal any compound-related effects. Evaluation at 3 months showed subtle generalized or

centrilobular changes in the liver (characterized as increased vacuolation with fat deposition), which was evident in all exposed groups and most prominent in mid- and high-dose female groups (Table 4-12). The most frequently detected change was characterized as a generalized vacuolation. Some evidence was found for an increase in severity of the generalized vacuolation with increasing exposure level, but the incidence of this lesion in the control mice was substantial, especially in females (Table 4-12). Incidences for centrilobular vacuolation were significantly increased only for the mid-dose female group. No other changes were found.

Using the results from this study to select doses for a chronic study, Kirschman et al. (1986) expressed the opinion that the mid-dose level (587 mg/kg-day) was the LOAEL in this study. Although incidences for generalized vacuolation were increased in the low- and mid-dose male groups, the incidences in the high-dose groups were not significantly increased compared with controls (Table 4-12). The study authors identified a LOAEL of 586 mg/kg-day for centrilobular vacuolation in male $B6C3F_1$ mice. The NOAEL for males was considered by the investigators to be between 226 and 587 mg/kg-day.

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Table 4-12. Incidences of histopathologic changes in livers of male and female B6C3F$_1$ mice exposed to dichloromethane in drinking water for 90 days

Lesion, by sex	Controls	Low dose	Mid dose	High dose
Males—n per group[a]	14	14	14	15
Estimated mean intake (mg/kg-day)	0	226	587	1,911
Number (%) with				
Hepatocyte vacuolation (generalized, centrilobular, or periportal)	9 (64)	12 (86)	13 (93)	12 (80)
Generalized vacuolation, severity:	7 (50)	12[b] (86)	13[b] (93)	10 (67)
minimal	4	3	9	7
mild	2	7	5	3
moderate	1	2	0	0
marked	0	0	0	0
Centrilobular severity:	2 (14)	0 (0)	1 (7)	5 (33)
minimal	2	0	0	1
mild	0	0	0	3
moderate	0	0	1	1
Females—n per group[a]	14	11	13	15
Estimated mean intake (mg/kg-day)	0	231	586	2,030
Number (%) with				
Hepatocyte vacuolation (generalized, centrilobular, or periportal)	13 (93)	11 (100)	13 (100)	13 (87)
Generalized vacuolation severity:	13 (93)	11 (100)	13 (100)	13 (87)
minimal	1	3	5	3
mild	8	7	6	6
moderate	4	1	2	1
marked	0	0	0	3
Centrilobular severity:	0 (0)	0 (0)	5[c] (39)	1 (7)
minimal	0	0	0	0
mild	0	0	2	1
moderate	0	0	3	0
marked	0	0	0	0

[a]20 per group; 5 sacrificed at 1 month.
[b]Statistical significance testing not reported by authors; Fisher's exact test for comparison with control p - value = 0.10 for low dose group and p = 0.032 for mid-dose group (two-sided).
[c]Statistical significance testing not reported by authors; Fisher's exact test for comparison with control p - value = 0.016 (two-sided). Authors say LOAEL = 587 mg/kg-day; NOAEL between 226 and 587 mg/kg-day for males; not explicitly stated for females.

Source: Kirschman et al. (1986).

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3549
3550 **4.2.1.2. *Toxicity Studies of Chronic Oral Exposures: Hepatic Effects and Carcinogenicity***

3551 Longer-term (up to 2-year) oral exposure studies in mice and rats are summarized in
3552 Table 4-13 and described in more detail below. These studies provide additional information
3553 pertaining to hepatotoxicity and carcinogenicity.

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Table 4-13. Studies of chronic oral dichloromethane exposures (up to 2 years)

Reference, strain/species	Number per group	Exposure information	Comments
Serota et al. (1986a) F344 rats	85/sex/dose + 135 controls	Drinking water, 2 years, target dose 0, 5, 50, 125, 250 mg/kg-day Mean intake: males 0, 6, 52, 125, 235 mg/kg-day females 0, 6, 58, 136, 263 mg/kg-day	Nonneoplastic liver effects (foci/areas of alteration) in males and females (see Table 4-14) Jagged stepped pattern of increasing incidence of neoplastic nodules or hepatcellular carcinoma in females (i.e., increased in the 50 and 250 mg/kg-day but not 125 mg/kg-day groups) (see Table 4-14)
Serota et al. (1986b); Hazelton Laboratories (1983) B6C3F$_1$ mice	Males 125, 200, 100, 100, 125 Females 100, 100, 50, 50, 50	Drinking water, 2 years, target dose 0, 60, 125, 185, 250 mg/kg-day Mean intake: males 0, 61, 124, 177, 234 mg/kg-day females 0, 59, 118, 172, 238 mg/kg-day	Increasing trend of liver cancer (hepatocellular adenoma or carcinoma) in males (see Table 4-15)
Maltoni et al. (1988) Sprague-Dawley rats	50/sex/dose	Gavage, up to 64 weeks 0, 100, 500 mg/kg-day, 4–5 days per week	High mortality in high dose group led to termination of study at 64 weeks; non-statistically significant increase in malignant mammary tumors in female rats
Maltoni et al. (1988) Swiss mice	50/sex/dose + 60 controls	Gavage, up to 64 weeks 0, 100, 500 mg/kg-day, 4–5 days per week	High mortality in high dose group led to termination of study at 64 weeks

4.2.1.2.1. ***Chronic oral exposure in F344 rats (Serota et al., 1986a).*** Treatment with dichloromethane did not induce adverse clinical signs or affect survival in the F344 rats (Serota et al., 1986a). BWs of rats in the 125 and 250 mg/kg-day groups were generally lower than in controls throughout the study. The authors stated that the differences, although small, were statistically significant, but the data were not shown in the published report. Water consumption was lower throughout the study in both sexes of rats from the 125 and 250 mg/kg-day groups relative to controls; food consumption was also lower in these groups during the first 13 weeks of treatment. Mean hematocrit, hemoglobin, and red blood cell count were increased in both sexes at dichloromethane levels of 50, 125, and 250 mg/kg-day for 52 and 78 weeks. Half of these increases were reported to be statistically significant, but the report did not provide the numerical values or specify which parameters were significant. Clinical chemistry results showed decreases in alkaline phosphatase (AP), creatinine, blood urea nitrogen, total protein, and cholesterol in both sexes at 250 mg/kg-day, and most of these changes were statistically

significant at one or both of the intervals evaluated. (Significant parameters not specified and the mean group values were not presented in the published report.) No significant deviations in urinary parameters were observed. Organ weights were not significantly affected by treatment with dichloromethane.

No treatment-related histopathological effects were noted in the tissues examined except for the liver (Serota et al., 1986a). Examination of liver sections showed a dose-related positive trend (positive Cochran-Armitage trend test) in the incidences of foci/areas of cellular alteration in treated F344 rats (Table 4-14). Comparisons of incidences with control incidences indicated statistically significant elevations at all dose levels except 5 mg/kg-day. These liver changes

Table 4-14. Incidences of nonneoplastic liver changes and liver tumors in male and female F344 rats exposed to dichloromethane in drinking water for 2 years

	Target dose (mg/kg-day)						
	Controls 0[a]	5	50	125	250	Trend *p*-value[b]	250 with recovery[c]
Males—n per group[d]	76	34	38	35	41		15
Estimated mean intake (mg/kg-day)	0	6	52	125	235		232
Number (%) with							
Liver foci/areas of alteration	52 (70)	22 (65)	35 (92)[e]	34 (97)[e]	40 (98)[e]	<0.0001	15 (100)[e]
Neoplastic nodules	9 (12)	1 (3)	0 (0)	2 (6)	1 (2)	Not reported	2 (13)
Hepatocellular carcinoma	3 (4)	0 (0)	0 (0)	0 (0)	1 (2)	Not reported	0 (0)
Neoplastic nodules and hepatocellular carcinoma	12 (16)	1 (3)	0 (0)	2 (6)	2 (5)	Not reported	2 (13)
Females—n per group[d]	67	29	41	38	34		20
Estimated mean intake (mg/kg-day)	0	6	58	136	263		239
Number (%) with							
Liver foci/areas of alteration	34 (51)	12 (41)	30 (73)[e]	34 (89)[e]	31 (91)[e]	<0.0001	17 (85)[e]
Neoplastic nodules	0 (0)	1 (3)	2 (5)	1 (3)	3 (9)	Not reported	2 (10)
Hepatocellular carcinoma	0 (0)	0 (0)	2 (5)	0 (0)	2 (6)	Not reported	0 (0)
Neoplastic nodules and hepatocellular carcinoma	0 (0)	1 (3)	4 (10)[f]	1 (3)	5 (14)[f]	$p < 0.01$	2 (10)[f]

[a]Two control groups combined.
[b]Cochran-Armitage trend test was used for trend test of liver foci/areas of alteration. For tumor mortality-unadjusted analysis, a Cochran-Armitage trend test was used, and, for tumor mortality-adjusted analyses, tumor prevalence analytic method by Dinse and Lagakos (1982) was used. Similar results were seen in these two analyses.
[c]Recovery group was exposed for 78 weeks and then had a 26-week period without dichloromethane exposure; n = 15 for nonneoplastic lesions and n = 17 for neoplastic lesions.
[d] Number available at terminal sacrifice; starting with 135 controls (combining both control groups) and 85 per sex per dose group except recovery group (n = 25); subtracted 5, 10, and 20 per group (except for recovery group) sacrificed at 25, 52, and 78 weeks, respectively, and subtracted unscheduled deaths, which ranged from 5 to 19 per group.
[e]Significantly ($p < 0.05$) different from control with Fisher's exact test.
[f]Significantly ($p < 0.05$) different from controls with Fisher's exact test, mortality-unadjusted and mortality-adjusted analyses.

Source: Serota et al. (1986a).

were first noted after treatment for 78 weeks and progressed until week 104. Livers of animals treated with dichloromethane also showed an increased incidence of fatty change, but incidence data for this lesion were not presented in the published report. The recovery group also showed an increased incidence of areas of cellular alterations, but the fatty changes were less pronounced than in the 250 mg/kg-day group dosed for 104 weeks. The authors indicate that 5 mg/kg-day was a NOAEL and 50 mg/kg-day was a LOAEL for nonneoplastic liver changes in male and female F344 rats exposed to dichloromethane in drinking water for 2 years.

Dichloromethane-exposed male rats showed no statistically significant increased incidence of liver tumors. In females, there was a positive trend for increasing incidence of hepatocellular carcinoma or neoplastic nodules with increasing dose (Table 4-14) (Serota et al., 1986a). Statistically significant increases in tumor incidences were observed in the 50 and 250 mg/kg-day groups (incidence rates of 10% and 14%, respectively) but not in the 125 mg/kg-day group (incidence rate of 3%). Incidence was also increased (10%) in a group exposed for 78 weeks followed by a 26-week period of no exposure. The characterization of malignant potential of the nodules was not described, however, and no trend was seen in the data limited to hepatocellular carcinomas. The incidence of hepatocellular carcinoma or neoplastic nodules in this control group (0%) was lower than that seen in historical controls from the same laboratory (324 female F344 rats; 4 with carcinoma, 21 with neoplastic nodules; 25/324 = 7.7%).

4.2.1.2.2. ***Chronic oral exposure in B6C3F$_1$ mice (Serota et al., 1986b; Hazelton Laboratories, 1983).*** A 2-year drinking water study similar to the previously described study in F344 rats was also conducted in B6C3F$_1$ mice (Serota et al., 1986b; Hazelton Laboratories, 1983). The mice received target doses of 0, 60, 125, 185, or 250 mg/kg-day of dichloromethane in deionized drinking water for 24 months. Treatment groups consisted of 100 female mice in the low-dose group and 50 in the remaining treatment groups. There were 200, 100, 100, and 125 male mice (low- to high-dose groups) in the treated groups. One hundred females (in two groups of 50) and 125 males (in two groups of 60 and 65 mice) served as controls. (The authors do not state why two groups of control mice were used, other than to say that the design was used due to the high and erratic incidence of liver tumors in historical control B6C3F$_1$ mice.) Based on water consumption and BW measurements, mean intakes were reported to be 61, 124, 177, and 234 mg/kg-day for males and 59, 118, 172, and 238 mg/kg-day for females. Endpoints examined included clinical signs, BW and water consumption, hematology at weeks 52 and 104, and gross and microscopic examinations of tissues and organs at termination. All tissues from the control and 250 mg/kg-day groups were examined microscopically, as well as the livers and neoplasms from all groups and the eyes of all males from all groups.

Throughout the 2-year study, mice from both control and treatment groups exhibited a high incidence of convulsions (Serota et al., 1986b; Hazelton Laboratories, 1983). The convulsions were noted only during handling for BW determinations, and efforts to establish a

basis for this response were unsuccessful. The incidence of convulsions did not correlate with an increased mortality rate. Survival to 104 weeks was high (82% in males and 78% in females), and no evidence for exposure-related negative effects on survival were found. Exposure had no significant effect on BW or water consumption. Mean leukocyte count was significantly elevated in males and females dosed with 250 mg/kg-day dichloromethane for 52 weeks, but the authors indicated that the mean values were within the normal historical range for the laboratory. Treatment-related nonproliferative histopathologic effects were restricted to the liver and consisted of a marginal increase in the amount of Oil Red O-positive material in the liver of males and females dosed with 250 mg/kg-day (group incidences for this lesion, however, were not presented in the published report). The results indicate that 185 mg/kg-day was a NOAEL and 250 mg/kg-day was a LOAEL for marginally increased amounts of fat in livers of male and female $B6C3F_1$ mice.

Incidences for proliferative hepatocellular changes in female mice were not presented in the published reports (Serota et al., 1986b; Hazelton Laboratories, 1983), but it was reported that exposed female mice did not show increased incidences of proliferative hepatocellular lesions. In the male $B6C3F_1$ mice, incidences for hepatic focal hyperplasia showed no evidence of an exposure-related effect (Table 4-15). The authors interpret the data regarding adenomas alone or carcinomas alone as showing no significantly elevated incidence compared with controls. The trend tests for each of these outcomes were 0.172 and 0.147 (Hazelton Laboratories, 1983), respectively, and none of the comparisons between individual exposure groups and the controls was statistically significant at the chosen Bonferroni-corrected level of <0.01. However, exposed male mice showed increased combined incidences of hepatocellular adenomas and carcinomas, with a linear trend *p*-value = 0.058 and individual *p*-values of <0.05.

Table 4-15. Incidences for focal hyperplasia and tumors in the liver of male $B6C3F_1$ mice exposed to dichloromethane in drinking water for 2 years

	Target dose (mg/kg-day)					
	Controls 0[a]	60	125	185	250	Trend *p*-value[b]
n per group[c]	125	200	100	99	125	
Estimated mean intake (mg/kg-day)	0	61	124	177	234	
Number (%) with						
Focal hyperplasia[d]	10 (8)	14 (7)	4 (4)	10 (10)	13 (10)	not reported
Hepatocellular adenoma	10 (8)	20 (10)	14 (14)	14 (14)	15 (12)	
mortality-adjusted percent and *p*-value[e]	(9)	(12)	(17)	(16)	(12)	
		p = 0.24	*p* = 0.064	*p* = 0.076	*p* = 0.13	0.172
Hepatocellular carcinoma	14 (11)	33 (17)	18 (18)	17 (17)	23 (18)	
mortality-adjusted percent and *p*-value[e]	(13)	(19)	(21)	(19)	(21)	
		p = 0.082	*p* = 0.073	*p* = 0.11	*p* = 0.044	0.147
Hepatocellular adenoma or carcinoma	24 (19)	51 (26)	30 (30)	31 (31)	35 (28)	0.058
mortality-adjusted percent and *p*-value[e]	(21)	(29)	(34)	(34)	(32)	
		p = 0.071	*p* = 0.023	*p* = 0.019	0.036	

[a]Two control groups combined.
[b]Cochran-Armitage trend test (source: Hazelton Laboratories [1983])
[c]Number at start of treatment.
[d]Some mice with hyperplasia also had hepatocellular neoplasms, but the exact number was unspecified by Serota et al. (1986b).
[e]Percent calculated based on number at risk, using Kaplan-Meier estimation, taking into account mortality losses; *p*-value for comparison with control group, using asymptotic normal test (source: Hazelton Laboratories [1983]).

Sources: Serota et al. (1986b); Hazelton Laboratories (1983).

Serota et al. (1986b) noted, in summary, that slight increases in proliferative hepatocellular lesions were found in exposed male but not female $B6C3F_1$ mice and that the increases were not dose related and were within the range of historical control incidences. The average incidence of hepatocellular adenomas and carcinomas in 354 control male $B6C3F_1$ mice in the laboratory in which the experiment was performed was 17.8% with a range of 5–40% (Serota et al., 1986b). Serota et al. (1986b) concluded that dichloromethane "did not induce a treatment-related carcinogenic response in $B6C3F_1$ mice" under the conditions of this study. An alternative conclusion, as determined by EPA, is that dichloromethane induced a carcinogenic response in male $B6C3F_1$ mice as evidenced by small, but statistically significant, increases in hepatocellular adenomas and carcinomas at dose levels of 125, 185, and 250 mg/kg-day but not at 60 mg/kg-day and by a marginally increased trend test for combined hepatocellular adenomas and carcinomas. Results for the highest dose group (no effect on BW, histologic findings restricted to mild histologic changes in the liver [vacuolation], and a slight, but statistically significant, increase in incidence in liver tumors in males only) indicate that this mouse study may not have included the maximum tolerated dose.

4.2.1.2.3. ***Chronic oral exposure in Sprague-Dawley rats and Swiss mice (Maltoni et al., 1988).*** Maltoni et al. (1988) conducted gavage carcinogenicity studies in Sprague-Dawley rats and in Swiss mice. Groups of rats (50/sex/dose level) were gavaged with dichloromethane (99.9% pure) in olive oil at dose levels of 0 (olive oil), 100, or 500 mg/kg-day, 4–5 days/week for 64 weeks. This dosing regime was also used for groups of Swiss mice (50/sex/dose level plus 60/sex as controls). Endpoints monitored included clinical signs, BW, and full necropsy at sacrifice (when spontaneous death occurred). For each animal sacrificed, histopathologic examinations were performed on the following organs: brain and cerebellum, zymbal glands, interscapular brown fat, salivary glands, tongue, thymus and mediastinal lymph nodes, lungs, liver, kidneys, adrenals, spleen, pancreas, esophagus, stomach, intestine, bladder, uterus, gonads, and any other organs with gross lesions. High mortality was observed in male and female high-dose rats (data not shown) and achieved significance ($p < 0.01$) in males. The increased mortality became evident after 36 weeks of treatment and led to the termination of treatment at week 64. Explanation of the mortality was not provided by the study authors. As with the rats,

high mortality occurred in male and female mice from the high-dose group ($p < 0.01$), and the exposure was terminated after 64 weeks.

Little information is provided regarding nonneoplastic effects (Maltoni et al., 1988). Treatment with dichloromethane did not affect BW in the Sprague-Dawley rats. A reduction in BW was apparent in treated mice after 36–40 weeks of treatment, but no data were shown to determine the magnitude of the effect. The lack of reporting of nonneoplastic findings from the histopathologic examinations precludes assigning NOAELs and LOAELs for possible nonneoplastic effects in these studies

The Maltoni et al. (1988) studies of Sprague-Dawley rats and Swiss mice did not find distinct exposure-related carcinogenic responses following gavage exposure to dichloromethane at dose levels up to 500 mg/kg-day, although the early termination of the study (at 64 weeks) limits the interpretation of this finding. Dichloromethane exposure was not related to the percentage of either study animal bearing benign and malignant tumors or bearing malignant tumors or to the number of total malignant tumors per 100 animals. High-dose female rats showed an increased incidence in malignant mammary tumors, mainly due to adenocarcinomas (8, 6, and 18% in the control, 100, and 500 mg/kg dose groups, respectively; the number of animals examined was not provided), but the increase was not statistically significant. A dose-related increase, although not statistically significant, in pulmonary adenomas was observed in male mice (5, 12, and 18% in control, 100, and 500 mg/kg-day groups, respectively). When mortality was taken into account, high-dose male mice that died in the period ranging from 52 to 78 weeks were reported to show a statistically significantly ($p < 0.05$) elevated incidence for pulmonary tumors (1/14, 4/21, and 7/24 in control, 100, and 500 mg/kg-day groups, respectively). Details of this analysis were not provided. EPA applied a Fisher's exact test to these incidences and determined a *p*-value of 0.11 for the comparison of the 500 mg/kg-day group (7/24) to the controls (1/14).

4.2.2. Inhalation Exposure: Overview of Noncancer and Cancer Effects

Inhalation dichloromethane exposure studies in rats and mice, using subchronic and chronic durations, identify the CNS, liver, and lungs as potential toxicity targets. Data from other studies indicate that hamsters are less susceptible to the nonneoplastic and neoplastic effects of dichloromethane than are rats and mice.

Increased incidences of nonneoplastic liver lesions were observed in Sprague-Dawley rats exposed to ≥500 ppm for 2 years (Nitschke et al., 1988a; Burek et al., 1984), F344 rats exposed to concentrations ≥1,000 ppm for 2 years (Mennear et al., 1988; NTP, 1986), and $B6C3F_1$ mice exposed to ≥2,000 ppm for 2 years (Mennear et al., 1988; NTP, 1986).

Two-year inhalation exposure studies at concentrations of 2,000 or 4,000 ppm dichloromethane produced increased incidences of lung and liver tumors in $B6C3F_1$ mice (Mennear et al., 1988; NTP, 1986). Additional studies examining mechanistic issues regarding

this effect are described in sections 4.5.2 and 4.5.3 (Maronpot et al., 1995; Foley et al., 1993; Kari et al., 1993). Significantly increased incidences of benign mammary tumors (primarily fibroadenomas) were also observed in male and female F344/N rats exposed by inhalation to 2,000 or 4,000 ppm for 2 years (Mennear et al., 1988; NTP, 1986). In the male rats, the incidence of fibromas or sarcomas originating from the subcutaneous tissue around the mammary gland was also increased, but the difference was not statistically significant. In other studies in Sprague-Dawley rats with exposures of 50–500 ppm (Nitschke et al., 1988a) and 500–3,500 ppm (Burek et al., 1984), the incidence of benign mammary tumors was not increased, but in females the number of tumors per tumor-bearing rat increased at the higher dose levels.

No obvious clinical signs of neurological impairment were observed in the 2-year bioassays involving exposure concentrations up to 2,000 ppm in F344 rats (Mennear et al., 1988; NTP, 1986) or 3,500 ppm in Sprague-Dawley rats (Nitschke et al., 1988a; Burek et al., 1984). In $B6C3F_1$mice exposed to 4,000 ppm in $B6C3F_1$, there was some evidence of hyperactivity during the first year of the study and lethargy during the second year, with female mice appearing to be more sensitive (Mennear et al., 1988; NTP, 1986). Studies that evaluated batteries of neurobehavioral endpoints following subchronic or chronic inhalation exposure are restricted to one in which no effects were observed more than 64 hours postexposure in an observational battery, a test of hind-limb grip strength, a battery of evoked potentials, or histologic examinations of brain, spinal cord, or peripheral nerves in F344 rats exposed to concentrations up to 2,000 ppm for 13 weeks (Mattsson et al., 1990) (see section 4.2.3)

No effects on reproductive performance were found in a two-generation reproductive toxicity study with F344 rats exposed to concentrations up to 1,500 ppm for 14 and 17 weeks before mating of the F0 and F1 generations, respectively (Nitschke et al., 1988b) (described more completely in section 4.3). Developmental effects following exposure of Long-Evans rats to 4,500 ppm for 14 days prior to mating and during gestation (or during gestation alone) included decreased offspring weight at birth and changed behavioral habituation of the offspring to novel environments (Bornschein et al., 1980; Hardin and Manson, 1980) (see section 4.3 for more details). In standard developmental toxicity studies involving exposure to 1,250 ppm on GDs 6–15, no adverse effects on fetal development were found in Swiss-Webster mice or Sprague-Dawley rats (Schwetz et al., 1975) (see section 4.3).

4.2.2.1. *Toxicity Studies of Subchronic Inhalation Exposures: General, Renal, and Hepatic Effects*

Data pertaining to general (e.g., BW, mortality), hepatic, and renal effects from several inhalation exposure studies in various species, with exposure periods of 3–6 months, are described below. (Studies providing detailed neurological data are described separately in section 4.4.3.) The earliest study involved several different species, with exposures of 5,000 ppm for up to 6 months (Heppel et al., 1944). Two 14-week studies in dogs, monkeys,

rats, and mice were conducted with exposures at 0, 1,000, and 5,000 ppm (Haun et al., 1972, 1971; Weinstein et al., 1972) and at 0, 25, and 100 ppm (Haun et al., 1972). Neurological effects and hepatic degeneration were seen at the 1,000 ppm dose. In the lower-dose portion of the Haun et al. (1972) study in mice, decreased cytochrome P-450 levels in liver microsomes and some histopathologic liver changes (fat stains and cytoplasmic vacuolation) were seen at 100 ppm but more obvious adverse effects were not observed. Leuschner et al. (1984) reported data from a high exposure (10,000 ppm) 90-day study of rats; beagle dogs were also included in this study, at an exposure level of 5,000 ppm. No evidence of toxicity was reported by the authors of this study. In a 13-week exposure study conducted by NTP (1986), decreased BWs and increased incidence of foreign body pneumonia were seen at 8,400 ppm in F344 rats, and histologic changes in the liver in B6C3F1 mice were seen at 4,200 ppm.

The first experimental study of dichloromethane exposure included dogs, rabbits, guinea pigs, and rats, with an exposure of approximately 5,000 ppm for 7 hours/day, 5 days per week for up to 6 months (Heppel et al., 1944). The strains of the animals, the comparability between exposed and unexposed group (in terms of sex distribution and other attributes), and process by which animals were chosen for histologic examination are not clearly described in the report. Exposed animals included adult dogs (1 male and 5 females), juvenile dogs (1 male and 1 female born in the exposure chamber and exposed daily from birth), adult rabbits (2 males and 2 females), guinea pigs (14 males), and rats (15 males and 6 females). The nonexposed control group included 14 guinea pigs, 28 rats, 4 rabbits, and an unspecified number of dogs. Exposure produced no significant effects on BWs except in the guinea pigs; after 131 exposures, average BWs were 0.820 and 1.025 kg for exposed and control guinea pigs, respectively. Three exposed guinea pigs died after 35, 90, and 96 exposures. No other deaths occurred, except for one exposed female rat that died after 22 exposures and giving birth to a litter. Autopsy showed thrombi in the renal vessels associated with marked cortical infarction. No adverse clinical signs of toxicity (such as decreased activity or incoordination) were observed in exposed animals during the study. Urinalysis, hematology tests, and tests of liver function performed on dogs during the study showed no treatment-related effects. At termination, gross and microscopic examination of the major organs showed no pathological changes after exposure to 5,000 ppm dichloromethane, with the exception that two of the exposed guinea pigs that died showed extensive pneumonia associated with moderate centrilobular fatty degeneration of the liver. The results indicate that 5,000 ppm was a NOAEL for nonneoplastic systemic effects in dogs, rabbits, and rats exposed 7 hours/day, 5 days/week for up to 6 months. The findings of three deaths (two with pulmonary congestion and centrilobular fatty degeneration) and 20% decreased average BW among the 14 exposed guinea pigs indicates that 5,000 ppm was a LOAEL in this species.

Haun et al. (1972, 1971) and Weinstein et al. (1972) reported results from studies in which groups of 8 female beagle dogs, 4 female rhesus monkeys, 20 male Sprague-Dawley rats, and 380 female ICR mice were continuously exposed to 0, 1,000, or 5,000 ppm dichloromethane

for up to 14 weeks in whole-body exposure chambers. Gross and histopathologic examinations were scheduled to be made on animals that died or were sacrificed during or at termination of the study. At 5,000 ppm, obvious nervous system effects (e.g., incoordination, lethargy) were observed in dogs, monkeys, and mice. At 1,000 ppm, these effects were most apparent in dogs and monkeys (Haun et al., 1971). Food consumption was reduced in all species at 5,000 ppm and in dogs and monkeys at 1,000 ppm. All exposed animals either lost weight or showed markedly decreased BW gains compared with controls. For example, rats exposed to 1,000 or 5,000 ppm for 14 weeks showed average BWs that were roughly 10 and 20% lower than control values. Significant numbers of dogs (4) and mice (123), as well as 1 monkey, died within the first 3 weeks of exposure to 5,000 ppm. Because of this high mortality, all surviving 5,000 ppm animals were sacrificed at 4 weeks of exposure, except for one half (10) of the rats that went on to survive the 14-week exposure period. At 1,000 ppm, six of eight dogs died by 7 weeks, at which time the remaining two were sacrificed. Monkeys, rats, and all but a few mice survived exposure to 1,000 ppm for 14 weeks.

Gross examination of tissues showed yellow, fatty livers in dogs that died during exposure to 1,000 or 5,000 ppm, "borderline" liver changes in 3 monkeys exposed to 5,000 ppm, and mottled liver changes in 4/10 rats exposed to 5,000 ppm for 14 weeks (Haun et al., 1971). Comprehensive reporting of the histologic findings from this study were not available, but Haun et al. (1972) reported that the primary target organ was the liver and that in some exposed animals the kidney was also affected. Light and electron microscopy of liver sections from groups of 4–10 mice sacrificed after 1, 4, 8, and 12 hours and 1, 2, 3, 4, 6, and 7 days of exposure to 5,000 ppm showed hepatocytes with balloon degeneration (dissociation of polyribosomes and swelling of rough endoplasmic reticulum) as early as 12 hours of exposure (Weinstein et al., 1972). The degeneration peaked in severity after 2 days of exposure and, subsequently, partially reversed in severity. Information on possible histopathologic changes in mice exposed to 1,000 ppm was not provided.

The results from this study demonstrate that dogs and mice were more sensitive than were rats and monkeys to lethal effects, nervous system depression, and possibly liver effects from continuous exposure to 1,000 or 5,000 ppm. The results indicate that continuous exposure to 1,000 ppm was an adverse effect level for mortality and effects on the nervous system and liver in dogs (exposed for up to 4 weeks) and for BW changes in rats (exposed for 14 weeks). The 5,000 ppm level induced mortality in beagle dogs, ICR mice, and rhesus monkeys (but not in Sprague-Dawley rats); obvious nervous system effects in dogs, mice, monkeys, and rats; and gross liver changes in dogs, mice, monkeys, and rats.

Haun et al. (1972) also conducted studies with groups of 20 mice, 20 rats, 16 dogs, and 4 monkeys exposed continuously to 0, 25, or 100 ppm dichloromethane for 100 days (14 weeks). The animals presumably were of the same strains and sexes as those used in the studies involving exposure to 1,000 or 5,000 ppm dichloromethane (Haun et al., 1972, 1971; Weinstein et al.,

1972). All animals underwent necropsy and histopathologic evaluation at termination of the exposure, but a list of the tissues examined and incidence or severity data were not presented in the report. Hematology and clinical chemistry variables (including COHb levels) were measured in blood samples collected from dogs and monkeys at biweekly or monthly intervals during exposure. COHb levels were elevated in a dose-related manner in monkeys and peaked at about 5% (approximately 0.8% pre-exposure) after 6 weeks of exposure. COHb levels in dogs were unaffected by the 25 ppm exposure level and rose to about 2% (from about 0.6%) from week 4 on in high-dose dogs. Additional groups of mice were included for assessment of hexobarbital sleep times at monthly intervals; levels of cytochromes P-450, P-420, and b_5 in liver microsomes at monthly intervals; and spontaneous physical activity at several intervals during the study.

No clinical signs of toxicity or alterations in weight gain were seen in any of the species examined. In dogs and monkeys, hematology and clinical chemistry results throughout the study and at termination were unremarkable, as were the results of the gross and histopathologic examinations. In mice exposed to 100 ppm, CYP levels in liver microsomes were significantly decreased (compared with control values) after 30, 60, and 90 days of exposure to 100 ppm, whereas levels of cytochrome b_5 and P-420 decreased after 30 days and increased after 90 days of exposure. At 25 ppm, no significant differences from control were seen in mouse liver levels of cytochromes. Mice exposed to 25 ppm showed no histopathologic changes, while histologic changes in mice at 100 ppm were restricted to positive fat stains and some cytoplasmic vacuolation in the liver. In rats at both exposure levels, the livers showed positive staining for increased fat, and the kidneys showed evidence of nonspecific tubular degenerative and regenerative changes. Haun et al. (1972) indicate that no distinctively adverse effects were found in monkeys, dogs, rats, or mice continuously exposed to 25 or 100 ppm for up to 14 weeks. Decreased CYP levels in liver microsomes and some histopathologic liver changes (fat stains and cytoplasmic vacuolation) were seen at the 100 ppm dose.

Leuschner et al. (1984) exposed Sprague-Dawley rats (20/sex/dose level) to 0 or 10,000 ppm and beagle dogs (3/sex/dose level) to 0 or 5,000 ppm dichloromethane in whole-body exposure chambers. Exposure periods were 6 hours/day for 90 consecutive days. Endpoints evaluated in both species included clinical signs, food and water consumption, BW, hematology, clinical chemistry, urinalysis, and gross and microscopic evaluation of 27 organs at termination. Electrocardiography and blood pressure measurements were also done in dogs. The only significant effect observed in rats was a slight redness of the conjunctiva 1–10 hours after each exposure. In dogs, compound-related effects were restricted to slight sedation throughout the exposure period and slight erythema lasting up to 10 hours after exposure. In this 90-day study involving daily 6-hour exposures, 10,000 and 5,000 ppm were NOAELs for behavioral, clinical chemistry, hematologic, and histologic signs of toxicity in Sprague-Dawley rats and beagle dogs, respectively.

NTP (1986) exposed groups of F344 rats and B6C3F$_1$ mice (10/sex/dose level) to target concentrations of 0, 525, 1,050, 2,100, 4,200, or 8,400 ppm dichloromethane, 6 hours/day, 5 days/week for 13 weeks in whole-body exposure chambers. Endpoints monitored included clinical signs, BW, and necropsy at termination. Comprehensive sets of tissues and organs in control and high-dose animals were histologically examined; tissues from the lower dose groups were examined to determine the no-observed-effect level. One male and one female rat from the 8,400 ppm exposure group died before the end of the study, but the cause of death was not discussed. The final mean BWs of 8,400 ppm male and female rats were reduced by 23 and 11%, respectively, relative to controls. Foreign-body pneumonia was present in 4/10 male and 6/10 female rats exposed to 8,400 ppm and in 1/10 female rat from the 4,200 ppm exposure group. The liver lipid/liver weight ratios for 8,400 ppm rats of both sexes and 4,200 ppm female rats were significantly lower than in controls. In mice, 4/10 males and 2/10 females exposed to 8,400 ppm died before the end of the study, and these deaths were considered treatment related. Histologic changes in exposed mice consisted of hepatic centrilobular hydropic degeneration (of minimal to mild severity) in 3/10 males and 8/10 females at 8,400 ppm and in 9/10 females from the 4,200 ppm exposure group. Histologic changes in the 2,100 ppm mouse group were not mentioned. The liver lipid/liver weight ratio for the high-dose female mice was significantly lower than in controls. In this 13-week study involving 6-hour exposure periods for 5 days/week, 4,200 ppm was a NOAEL and 8,400 ppm was a LOAEL for decreased BWs and increased incidence of foreign-body pneumonia in F344 rats. In B6C3F$_1$ mice, 2,100 ppm was a NOAEL and 4,200 ppm was a LOAEL for histologic changes in the liver.

4.2.2.2. *Toxicity Studies from Chronic Inhalation Exposures*

Chronic inhalation exposure studies are summarized in Table 4-16. Details of each study are described below, with the results pertaining to nonneoplastic and neoplastic effects summarized in the following sections.

3920

Table 4-16. Studies of chronic inhalation dichloromethane exposures

Reference, strain/species	Number per group	Exposure information	Comments
Mennear et al. (1988); NTP (1986) F344 rats	50/sex/dose	2 years, 6 hours/day, 5 days/week 0, 1,000, 2,000, 4,000 ppm	Nonneoplastic liver effects and hemosiderosis in males and females (see Table 4-17) Weak trend for neoplastic nodule or hepatocellular carcinoma in females, benign mammary tumors in males and females (see Table 4-18)
Mennear et al. (1988); NTP (1986) B6C3F$_1$ mice	50/sex/dose	2 years, 6 hours/day, 5 days/week 0, 2,000, 4,000 ppm	Varied nonneoplastic effects (see Table 4-19) Liver and lung tumors (adenomas or carcinomas) in males and females (see Table 4-20)
Burek et al. (1984) Syrian hamsters	95/sex/dose	2 years, 6 hours/day, 5 days/week 0, 500, 1,500, 3,500 ppm	Decreased mortality Increased CoHb at 500 ppm (see section 4.2.2.2.3)
Burek et al. (1984) Sprague-Dawley rats	92–97/sex/dose	2 years, 6 hours/day, 5 days/week 0, 500, 1,500, 3,500 ppm	Nonneoplastic liver effects in males and females (see Table 4-21) Increased CoHb at 500 ppm Increased number of benign mammary tumors per tumor bearing rat (females) (see Table 4-21)
Nitschke et al. (1988a) Sprague-Dawley rats	90/dose/sex	2 years, 6 hours/day, 5 days/week 0, 50, 200, 500 ppm	Nonneoplastic liver effects in males and females (statistically significant in females) (seeTable 4-22) Increased CoHb at 50 ppm Increased number of benign mammary tumors per animal in females (see Table 4-23)
Maltoni et al. (1988) Sprague-Dawley rats, female	54–60/dose	2 years, 4 hours/day, 5 days/week for 7 weeks; 7 hours/day, 5 days/week for 97 weeks 0, 100 ppm	No effects seen on total number of benign or malignant cancers

3921
3922

3923 **4.2.2.2.1.** ***Chronic inhalation exposure in F344/N rats (Mennear et al., 1988; NTP, 1986).***
3924 NTP conducted a 2-year inhalation exposure study in F344/N rats (Mennear et al., 1988; NTP,
3925 1986). The rats (50/sex/exposure level) were exposed to dichloromethane (>99% pure) by
3926 inhalation in exposure chambers, 6 hours/day, 5 days/week for 2 years. Exposure concentrations
3927 were 0, 1,000, 2,000, or 4,000 ppm. Mean daily concentrations never exceeded 110% of target
3928 and were <90% of target in only 23 of 1,476 analyses. Endpoints monitored included clinical
3929 signs, mortality, and gross and microscopic examinations of 32 tissues at study termination.
3930 Clinical examinations were conducted weekly for 3.5 months and biweekly until month 8. After
3931 8 months, the animals were clinically examined and palpated for tumors and masses monthly
3932 until the end of the study.
3933 Dichloromethane exposure did not significantly alter BW gain or terminal BWs
3934 (Mennear et al., 1988; NTP, 1986). Survival of male rats was low in all exposed groups and the

control group, and no significant exposure-related differences were apparent. Most deaths occurred during the last 16 weeks of the study. Survival at week 86 was 36/50, 39/50, 37/50, and 33/50 for the control, 1,000, 2,000, and 4,000 ppm groups, respectively. In female rats, there was a trend towards decreased survival, and the survival of high-dose female rats was significantly reduced, possibly due to leukemia. Survival in the females at 86 weeks was 30/50, 22/50, 22/50, and 15/50 for the control, 1,000, 2,000, and 4,000 ppm groups, respectively. Nonneoplastic lesions with statistically significantly elevated incidences, compared with controls, included hepatocyte cytoplasmic vacuolation and necrosis and liver hemosiderosis in males and females; renal tubular cell degeneration in males and females; splenic fibrosis in males; and nasal cavity squamous metaplasia in females (Table 4-17). The results indicate that 1,000 ppm (6 hours/day, 5 days/week) was a LOAEL for nonneoplastic liver changes (hepatocyte cytoplasmic vacuolation and necrosis, hepatic hemosiderosis) in male and female F344/N rats. A NOAEL was not established because effects were observed at the lowest dose.

3949

Table 4-17. Incidences of nonneoplastic histologic changes in male and female F344/N rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm)[a]			
	Controls			
Lesion, by sex	0	1,000	2,000	4,000
Males				
n per group[b]	50	50	50	50
Number (%)[c] with				
Liver changes				
Hepatocyte cytoplasmic vacuolation	8 (16)	26 (53)[d]	22 (44)[d]	25 (50)[d]
Hepatocyte focal necrosis	7 (14)	23 (47)[d]	6 (12)	16 (32)[d]
Hepatocytomegaly	2 (4)	10 (20)	6 (12)	5 (10)
Hemosiderosis	8 (16)	29 (59)[d]	37 (74)[d]	42 (84)[d]
Bile duct fibrosis	8 (16)	10 (20)	17 (34)	23 (46)[d]
Renal tubular cell degeneration	11 (22)	13 (26)	23 (46)[d]	10 (20)[d]
Splenic fibrosis	2 (4)	6 (12)	11 (22)[d]	8 (16)[d]
Females				
n per group[e]	50	50	50	50
Number (%)[c] with				
Liver changes				
Hepatocyte cytoplasmic vacuolation	10 (20)	43 (86)[d]	44 (88)[d]	43 (86)[d]
Hepatocyte focal necrosis	2 (4)	32 (64)[d]	19 (38)[d]	9 (18)[d]
Hepatocytomegaly	3 (6)	10 (20)[d]	18 (36)[d]	5 (10)
Hemosiderosis	19 (38)	29 (58)[d]	38 (76)[d]	45 (90)[d]
Bile duct fibrosis	4 (8)	3 (6)	10 (20)[d]	3 (6)
Renal tubular cell degeneration	14 (28)	20 (40)	22 (44)	25 (51)[d]
Splenic fibrosis	0 (0)	2 (4)	4 (8)	4 (8)
Nasal cavity squamous metaplasia	1 (2)	2 (4)	3 (6)	9 (18)[d]

[a]1,000 ppm = 3,474 mg/m^3, 2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.
[b]Number of male rats necropsied per group; only 49 1,000 ppm livers were examined microscopically.
[c]Percentages were based on the number of tissues examined microscopically per group.
[d]Statistical significance not reported in publications but significantly ($p \leq 0.05$) different from control as calculated by Fisher's exact test.
[e]Number of females necropsied per group; only 49 4,000 ppm kidneys and spleens were examined microscopically.

Sources: Mennear et al. (1988); NTP (1986, Appendix B, Tables C1 and C2).

3950
3951 Incidences of mammary fibroadenomas were significantly increased in 4,000 ppm males
3952 and 2,000 and 4,000 ppm females, compared with controls (Table 4-18). Similar patterns were
3953 seen with the combination of fibroadenomas and adenomas (not shown in Table 4-18). In males,
3954 subcutaneous tissue fibroma or sarcoma was seen in 1/50, 1/50, 2/50, and 5/50 rats in the 0,
3955 1,000, 2,000, and 4,000 ppm groups, respectively, but these lesions were not seen in females.
3956 Incidences of female rats with liver neoplastic nodules or carcinomas (combined) showed a
3957 significant trend test after survival adjustment only, but the incidences at the two highest dose
3958 levels were not significantly increased relative to the control (Table 4-18).
3959 Incidences for mononuclear cell leukemias in mid- and high-dose female rats were
3960 statistically significant after a survival-adjustment analysis. However, Mennear et al. (1988)
3961 considered the relationship between exposure to dichloromethane and mononuclear cell leukemia

to be equivocal, based on the fact that most male rats had leukemia (34/50, 26/50, 32/50, and 35/50 in controls, 1,000, 2,000, and 4,000 ppm rats, respectively). Other neoplasms that had increased incidences included mesotheliomas (predominantly in the tunica vaginalis) in males (0/50, 2/50, 5/50, and 4/50 in controls, 1,000, 2,000, and 4,000 ppm rats, respectively). This lesion was not considered to be related to dichloromethane exposure, because the concurrent control incidence (0/50) for this neoplasm was low relative to earlier inhalation studies conducted at this laboratory (4/100, 4%) and in other NTP studies with male F344/N rats (44/1,727) (mean historical percentage across NTP studies = 3 ± 2%).

NTP (1986) concluded that there was "some evidence of carcinogenicity of dichloromethane" in male F344/N rats as shown by increased incidence of benign mammary gland tumors and "clear evidence of carcinogenicity" of dichloromethane in female F344/N rats as shown by increased incidence of benign mammary gland tumors. The summary of the hepatic effects in rats in the NTP (1986) report also notes the positive trend in the incidence of hepatocellular neoplastic nodules or carcinomas in females, which "may have been due to dichloromethane exposure."

Table 4-18. Incidences of selected neoplastic lesions in male and female F344/N rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

Neoplastic lesion, by sex	Exposure (ppm) [a]												
	0 (Controls)			1,000			2,000			4,000			
	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	Trend *p*-value[d]
Males													
n per group	50			50			50			50			
Liver—Neoplastic nodule or hepatocellular carcinoma	2	(4)	(10)	3	(6)	(13)	4	(8)	(19)	1	(2)	(6)	0.55
Liver—hepatocellular carcinoma	2	(4)	(10)	1	(2)	(4)	2	(4)	(10)	1	(2)	(6)	nr
Lung—Bronchoalveolar adenoma or carcinoma	1			1	(2)		2	(4)		1	(2)		
Mammary gland													
Adenoma, adenocarcinoma, or carcinoma	0	(0)		0	(0)		0	(0)		1	(2)		
Subcutaneous tissue fibroma or sarcoma	1	(2)	(6)	1	(2)	(6)	2	(4)	(9)	5	(10)	(23)	0.008
Fibroadenoma	0	(0)	(0)	0	(0)	(0)	2	(4)	(12)	1	(2)	(8)	< 0.001
Mammary gland or subcutaneous tissue adenoma, fibroadenoma, fibroma, or sarcoma	1	(2)	(6)	1	(2)	(6)	4	(8)	(21)	9[e]	(18)	(49)	<0.001
Brain (carcinoma, not otherwise specified, invasive)	0	(0)		1	(2)		0	(0)		0	(0)		
Females													
n per group	50			50			50			50			
Liver—Neoplastic nodule or hepatocellular carcinoma	2	(4)	(7)	1	(2)	(2)	4	(8)	(14)	5	(10)	(20)	0.08
Liver—hepatocellular carcinoma	0	(0)	(0)	0	(0)	(0)	1	(2)	(4)	0	(0)	(0)	nr
Lung—Bronchoalveolar adenoma or carcinoma	1	(2)		1	(2)		0	(0)		0	(0)		
Mammary gland													
Adenocarcinoma or carcinoma	1	(2)		2	(4)		2	(4)		0	(0)		
Adenoma, adenocarcinoma, or carcinoma	1	(2)		2	(4)		2	(4)		1	(2)		
Fibroadenoma	5	(10)	(16)	11[e]	(22)	(41)	13[e]	(26)	(44)	22[e]	(44)	(79)	<0.001
Mammary gland adenoma, fibroadenoma, or adenocarcinoma	6	(12)	(18)	13	(26)	(44)	14[e]	(28)	(45)	23[e]	(46)	(86)	<0.001
Brain (carcinoma, not otherwise specified, invasive, and oligodendroglioma)[f]	1	(2)		0	(0)		2	(4)		0	(0)		

[a]1,000 ppm = 3,474 mg/m^3, 2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.
[b]Percentages based on the number of tissues examined microscopically per group; for males, 49 livers and lungs were examined microscopically in the 1,000 ppm groups, and only 49 brains were examined microscopically in the 4,000 ppm group. For comparison, incidence in historical controls reported in NTP (1986) were 1% for female liver tumors and 16% for female mammary fibroadenomas.
[c]Mortality-adjusted percentage.
[d]Life-table trend test, as reported by NTP (1986). nr = not reported.
[e]Life-table test comparison dose group with control < 0.05, as reported by NTP (1986).
[f]The oligodendroglioma occurred in the 2,000 ppm group.

Sources: Mennear et al. (1988); NTP (1986, Appendix A and Appendix E, Tables E1 and E2)

4.2.2.2.2. *Chronic inhalation exposure in B6C3F$_1$ mice (Mennear et al., 1988; NTP, 1986).* A 2-year inhalation exposure study in B6C3F$_1$ mice, similar to that in F344/N rats, was also conducted by NTP (Mennear et al., 1988; NTP, 1986). The mice (50/sex/exposure level) were exposed to dichloromethane (>99% pure) by inhalation at concentrations of 0, 2,000, or 4,000 ppm in exposure chambers 6 hours/day, 5 days/week for 2 years. As with the study in rats, mean daily concentrations in the mice never exceeded 110% of target and were <90% of target in only 23 of 1,476 analyses. Endpoints monitored included clinical signs, mortality, and gross and microscopic examinations of 32 tissues at study termination. Clinical examinations were conducted weekly for 3.5 months and biweekly until month 8. After 8 months, the animals were clinically examined and palpated monthly for tumors and masses until the end of the study.

The BW of 4,000 ppm males was comparable to controls until week 90 and 8–11% below controls thereafter. The BW of 4,000 ppm females was 0–8% lower than that of controls from week 51 to 95 and 17% lower at study termination. No information was provided regarding food consumption during the study. Male and female mice from the high-dose groups (4,000 ppm) were hyperactive during the first year of the study; during the second year, high-dose females appeared lethargic. Exposure was associated with decreased survivability of both male and female mice (males: 39/50, 24/50, and 11/50 and females: 25/50, 25/50, and 8/50 in controls, 2,000 ppm, and 4,000 ppm at 104 weeks, respectively). In 4,000 ppm mice, statistically significant incidences of nonneoplastic lesions were found in the liver (cytologic degeneration), testes (atrophy), ovary and uterus (atrophy), kidneys (tubule casts in males only), stomach (dilatation), and spleen (splenic follicles in males only) (Table 4-19). In 2,000 ppm mice, the only nonneoplastic lesions showing statistically significantly elevated incidences were ovarian atrophy, renal tubule casts, and hepatocyte degeneration in female mice (Table 4-19). The results indicate that 2,000 ppm, the lowest exposure level, was a LOAEL for nonneoplastic changes in the ovaries, kidneys, and livers of female B6C3F$_1$ mice. A NOAEL was not established because effects occurred at the lowest exposure level.

4008

Table 4-19. Incidences of nonneoplastic histologic changes in B6C3F$_1$ mice exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

Lesion, by sex	Exposure (ppm) [a]		
	Controls 0	2,000	4,000
Males: n per group[b]	50	50	50
Number (%)[c] with			
Liver changes			
Hepatocyte cytoplasmic vacuolation	Not reported	Not reported	Not reported
Hepatocyte focal necrosis	0 (0)	0 (0)	2 (4)
Cytologic degeneration	0 (0)	0 (0)	22 (45)[d]
Testicular atrophy	0 (0)	4 (8)	31 (62)[d]
Renal tubule casts	6 (12)	11 (22)	20 (40)[d]
Stomach dilatation	3 (6)	7 (15)	9 (18)[d]
Splenic follicular atrophy	0 (0)	3 (6)	7 (15)[d]
Females: n per group[e]	50	50	50
Number (%)[c] with			
Liver changes			
Hepatocyte cytoplasmic vacuolation	Not reported	Not reported	Not reported
Hepatocyte focal necrosis	Not reported	Not reported	Not reported
Cytologic degeneration	0 (0)	23 (48)[d]	21 (44)[d]
Ovarian atrophy	6 (12)	28 (60)[d]	32 (74)[d]
Uterus atrophy	0 (0)	1 (2)	8 (17)[d]
Renal tubule casts	8 (16)	23 (48)[d]	23 (49)[d]
Glandular stomach dilatation	1 (2)	2 (4)	10 (20)[d]
Splenic follicular atrophy	0 (0)	0 (0)	1 (2)

[a]2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.
[b]Number of male mice necropsied per group. The number biopsied in the 0, 2,000, and 4,000 ppm dose groups was 50, 49, and 49 for liver; 50, 49, and 50 for renal tubules; 49, 47, and 49 for stomach; and 49, 49, and 48 for spleen.
[c]Percentages were based on the number of tissues examined microscopically per group.
[d]Statistical significance not reported in publications but significantly different ($p \leq 0.05$) from control as calculated by EPA using Fisher's exact test.
[e]Number of females necropsied per group. The number biopsied in the 0, 2,000, and 4,000 ppm dose groups was 50, 48, and 48 for liver; 50, 47, and 43 for ovaries; 50, 48, and 47 for uterus; 49, 48, and 47 for renal tubule; 49, 47, and 48 for stomach; and 49, 48, and 47 for spleen.

Sources: Mennear et al. (1988); NTP (1986, Appendix C, Tables D1 and D2).

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4011 At both exposure levels, statistically significantly elevated incidences were found for
4012 hepatocellular adenomas (males only); hepatocellular carcinomas; hepatocellular adenomas and
4013 carcinomas, combined; bronchoalveolar adenomas; bronchoalveolar carcinomas; and
4014 bronchoalveolar adenomas and carcinomas (Table 4-20). Statistically significant positive trend
4015 tests were found for each of these tumor types in female mice. The trend tests were significant
4016 for the liver tumors in male mice after life-table adjustment for reduced survival. The only other
4017 statistically significant carcinogenic response was for increased incidence of hemangiosarcomas
4018 or combined hemangiomas and hemangiosarcomas in male mice exposed to 4,000 ppm. NTP

(1986) concluded that the elevated incidences of liver and lung tumors provided clear evidence of carcinogenicity in male and female $B6C3F_1$ mice.

Table 4-20. Incidences of neoplastic lesions in male and female $B6C3F_1$ mice exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm)[a]									
	0 (Controls)			2,000			4,000			
Neoplastic lesion, by sex	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	Trend *p*-value[d]
Males										
Liver										
Hepatocellular adenoma	10	(20)	(23)	14	(29)	(47)	14	(29)	(68)	0.19
Hepatocellular	13	(26)	(30)	15	(30)	(44)	26[e]	(53)	(76)	0.004
Hepatocellular adenoma or carcinoma	22	(44)	(48)	24	(49)	(67)	33[e]	(67)	(93)	0.013
Lung										
Bronchoalveolar adenoma	3	(6)	(8)	19[e]	(38)	(56)	24[e]	(48)	(79)	<0.001
Bronchoalveolar carcinoma	2	(4)	(5)	10[e]	(20)	(34)	28[e]	(56)	(93)	<0.001
Bronchoalveolar adenoma or carcinoma	5	(10)	(12)	27[e]	(54)	(74)	40[e]	(80)	(100)	<0.001
Mammary adenocarcinoma[f]	–			–			–			
Hemangioma or hemangiosarcoma, combined	2	(4)	(5)	2	(4)	(8)	6	(12)	(26)	0.08
Females										
Liver										
Hepatocellular adenoma	2	(4)	(7)	6	(13)	(21)	22[e]	(46)	(83)	<0.001
Hepatocellular carcinoma	1	(1)	(4)	11	(23)	(34)	32[e]	(67)	(97)	<0.001
Hepatocellular adenoma or carcinoma	3	(6)	(10)	16[e]	(33)	(48)	40[e]	(83)	(100)	<0.001
Lung										
Bronchoalveolar adenoma	2	(4)	(7)	23[e]	(48)	(58)	28[e]	(58)	(91)	<0.001
Bronchoalveolar carcinoma	1	(1)	(4)	13[e]	(27)	(46)	29[e]	(60)	(92)	<0.001
Bronchoalveolar adenoma or carcinoma	3	(6)	(11)	30[e]	(63)	(83)	41[e]	(85)	(100)	<0.001
Mammary adenocarcinoma	2	(4)	(8)	3	(6)	(10)	0	(0)	(0)	0.21
Hemangioma or hemangiosarcoma, combined[f]	–			–			–			

[a]2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.
[b]Percentages based on the number of tissues examined microscopically per group; for males, 49 livers were examined in the 2,000 and 4,000 ppm groups; for females, only 48 livers and lungs and 49 mammary glands were microscopically examined in the 2,000 and 4,000 ppm groups. For comparison, incidence in historical controls reported in NTP (1986) were 28% for male liver tumors, 31% for male lung tumors, 5% for female liver tumors, and 10% for female lung tumors.
[c]Mortality-adjusted percentage.
[d]Life-table trend test, as reported by NTP (1986).
[e]Life-table test comparison dose group with control < 0.05, as reported by NTP (1986).
[f]Data not reported.

Sources: Mennear et al. (1988); NTP, (1986, Appendix E, Tables E3 and E4).

4.2.2.2.3. *Chronic inhalation exposure in Syrian hamsters (Burek et al., 1984).* Burek et al. (1984) conducted a chronic toxicity and carcinogenicity study in rats and hamsters. In the hamster study, groups of 95 Syrian golden hamsters of each sex were exposed to 0 (filtered air), 500, 1,500, or 3,500 ppm dichloromethane (>99% pure) under dynamic airflow conditions in whole-body exposure chambers 6 hours/day, 5 days/week for 2 years. Exposure started when the

animals were approximately 8 weeks of age. Interim sacrifices were conducted at 6, 12, and 18 months. The hamsters were observed daily during exposure days and were palpated monthly for palpable masses starting the third month of the study. BWs were monitored weekly for the first 8 weeks of the study and monthly thereafter. Hematologic determinations included packed cell volume, total erythrocyte counts, total red blood cells, differential leukocyte counts, and hemoglobin concentration. The mean corpuscular volume, mean corpuscular hemoglobin, and MCHC were also determined. A reticulocyte count was also performed on all animals at the 18-month kill and on 10 animals/sex/dose at 24 months. Clinical chemistry determinations included serum AP and ALT activities, blood urea nitrogen levels, and total protein and albumin. Urinary parameters measured were specific gravity, pH, glucose, ketones, bilirubin, occult blood, protein, and urobilinogen. Hematology, clinical chemistries, and urinalysis were performed at interim sacrifices and at termination. COHb was measured after a single 6-hour exposure and following 22 months of exposure. Gross and microscopic examinations were conducted on all tissues. In addition, the weights of the brain, heart, liver, kidneys, and testes were recorded.

In the study using Syrian hamsters (Burek et al., 1984), hamsters were exposed to analytical concentrations of dichloromethane of 510 ± 27, 1,510 ± 62, and 3,472 ± 144 ppm for the target concentrations of 500, 1,500, and 3,500 ppm, respectively. No exposure-related clinical signs were observed in the hamsters throughout the study. Significantly decreased mortality was observed in females exposed to 3,500 ppm from the 13th through the 24th month and from the 20th to the 24th month in females exposed to 1,500 ppm. Exposure to dichloromethane had no significant effect on BW or on mean organ weights. Regarding hematology parameters (actual data were not shown), Burek et al. (1984) stated that a few statistically significant changes occurred, but no obvious pattern could be discerned and most values were within the expected range for the animals. There were no exposure-related alterations in clinical chemistry or urinalysis values. Male and female hamsters in all dose groups had significantly elevated COHb values after a single 6-hour exposure and after 22 months of exposure, but at both time points there was no dose-response relationship above the first dose level and no apparent significant differences in the magnitude of the changes between the two time points. For example, mean values (± SD) for percentage COHb in male hamsters after 22 months of exposure were 3.3 (± 3.5), 28.4 (± 5.9), 27.8 (± 2.9), and 30.2 (± 4.9), for the control through 3,500 ppm groups, respectively. Similar values were obtained for females at 22 months and for males and females after the first day of exposure. Pathological evaluation of hamsters showed a lack of evidence of definite target organ toxicity. Specific observations mentioned by the authors included a trend of increasing hemosiderin in the liver of male hamsters at 6 and 12 months; decreased amyloid deposit in organs, such as the liver, kidneys, adrenal, and thyroid glands in exposed animals; and fewer biliary cysts in the liver. Increased hepatic hemosiderin at the 12 month sacrifice was observed in 1/5, 1/5, 3/5, and 5/5 male hamsters in the control through 3,500 ppm groups, respectively. No exposure-related increased

incidences of hepatic hemosiderin, or other liver effects, were reported for the terminal sacrifice. The exposure-related decreases in geriatric changes (i.e., amyloid deposits and biliary cysts) were more prominent in females and were associated with the increased survivability in the exposed female hamsters compared with controls. The results indicate that 3,500 ppm was a NOAEL for adverse changes in clinical chemistry and hematological variables, as well as for nonneoplastic histologic changes in tissues, in male and female Syrian golden hamsters. A LOAEL was not established, based on the lack of adverse changes in clinical chemistry and hematological variables as well as the absence of nonneoplastic histologic changes in tissues, in male and female Syrian golden hamsters.

Evaluation of the total number of hamsters with a tumor, the number with a benign tumor, or the number with a malignant tumor revealed no exposure-related differences in male hamsters. In the high-dose female group, there was a statistically significant increase in the total number of benign tumors at any tissue site (the report did not specify which sites), but this was considered to be secondary to the increased survival of this group. Incidences of male or female hamsters with tumors in specific tissues were not statistically significantly elevated in exposed groups compared with control incidences. The results indicate that no statistically significant, exposure-related carcinogenic responses occurred in male or female Syrian golden hamsters exposed (6 hours/day, 5 days/week) to up to 3,500 ppm dichloromethane for 2 years.

4.2.2.2.4. ***Chronic inhalation exposure in Sprague-Dawley rats (Burek et al., 1984).*** In the rat study, groups of 92–97 Sprague-Dawley rats of each sex were exposed (similar to the hamster study described in the previous section) to 0, 500, 1,500, or 3,500 ppm dichloromethane 6 hours/day, 5 days/week for 2 years (Burek et al., 1984). Rats were approximately 8 weeks old when exposure started. Interim sacrifices were conducted at 6, 12, 15, and 18 months. Endpoints monitored in rats were the same as in hamsters except that total protein and albumin in blood were not determined in rats. In addition to measurement at scheduled sacrifices, serum ALT activity was also measured after 30 days of exposure. COHb was measured after 6, 11, 18, and 21 months of exposure. Bone marrow cells were collected for cytogenetic studies from five rats/sex/dose after 6 months of exposure. The scope of the pathological examinations of the rats was the same as in the hamster study.

No significant exposure-related signs of toxicity were observed in the rats during the study. A significant increase in mortality was seen in high-dose female rats from the 18th to the 24th month of exposure, and this appeared to be exposure-related. Exposure to dichloromethane had no significant effect on BW gain in either males or females. The only exposure-related alterations in organ weights was a significant increase in both absolute and relative liver weight in high-dose males at the 18-month interim kill and a significant increase in relative liver weight in high-dose females also at 18 months. Statistically significant changes in hematologic parameters were restricted to increased mean corpuscular volume and mean corpuscular

hemoglobin values at 15 months in males. The clinical chemistry tests revealed no significant exposure-related effects. Male and female rats in all exposed groups had significantly elevated COHb values at all time points, but no dose-response relationship was apparent. For example, mean (± SD) values for percentage COHb after 21 months of exposure were 0.4 (± 0.7), 12.8 (± 2.6), 14.8 (± 4.4), and 12.2 (± 5.7) for the control through 3,500 ppm female rat groups, respectively. Exposure-related statistically significant increases in incidences of nonneoplastic lesions were restricted to the liver (Table 4-21). The incidences of males or females with hepatocellular vacuolation consistent with fatty change increased as the exposure concentration increased. Hepatocellular necrosis occurred at elevated incidences in male rats exposed to 1,500 or 3,500 ppm, compared with controls, but this endpoint was not reported in the female data. Liver lesions were initially observed after 12 months of treatment. There was some evidence that exposure at the two highest levels provided some inhibition of the age-related glomerulonephropathy observed in the control rats at termination. The results indicate that the lowest exposure level, 500 ppm, was a LOAEL for fatty changes in the liver of male and female Sprague-Dawley rats and that exposure to ≥1,500 ppm induced hepatocellular necrosis in males.

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Table 4-21. Incidences of selected neoplastic histologic changes in male and female Sprague-Dawley rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm)[a]			
	Controls			
Lesion, by sex	0	500	1,500	3,500
Males—n per group	92	95	95	97
Number (%) with				
Liver changes				
Hepatocellular necrosis	2[b] (2)	8 (8)	10 (10)[c]	11 (11)[c]
Coagulation necrosis	--[d]	--	--	--
Hepatic vacuolation (fatty change)	16[b] (17)	36 (38)[c]	43 (45)[c]	52 (54)[c]
Foci of altered hepatocytes	--	--	--	--
Foci of altered hepatocytes, basophilic	--	--	--	--
Area of altered hepatocytes	--	--	--	--
Multinucleated hepatocytes	--	--	--	--
Glomerulonephropathy				
Severe	70[b] (76)	62 (65)	53 (56)[c]	39 (40)[c]
Any degree	92[b,e] (100)	91 (96)	93 (98)	90 (93)
Mammary changes				
Rats with benign mammary tumors	7[b] (8)	3 (3)	7 (7)	14 (14)
Total number of benign mammary tumors	8	6	11	17
Number of tumors per tumor-bearing rat[f]	1.1	2.0	1.6	1.2
Females—n per group	96	95	96	97
Number (%) with				
Liver changes				
Hepatocellular necrosis	–	–	–	–
Coagulation necrosis	1[b] (1)	0 (0)	2 (2)	7 (7)
Hepatic vacuolation (fatty change)	33[b] (34)	49 (52)[c]	56 (58)[c]	63 (65)[c]
Foci of altered hepatocytes	35[b] (37)	36 (38)	27 (28)	50 (52)[c]
Foci of altered hepatocytes, basophilic	3[b] (3)	0 (0)	4 (4)	10 (10)
Area of altered hepatocytes	19[b] (20)	24 (25)	28 (29)	35 (36)[c]
Multinucleated hepatocytes	7[b] (7)	36 (38)[c]	34 (35)[c]	29 (30)[c]
Glomerulonephropathy				
Severe	5 (5)	3 (3)	4 (4)	5 (5)
Any degree	62[b] (65)	64 (67)	59 (62)	48 (50)[c]
Mammary changes				
Rats with benign mammary tumors	79 (82)	81 (85)	80 (83)	83 (86)
Total number of benign mammary tumors	165	218	245	287
Number of tumors per tumor-bearing rat[f]	2.1	2.7	3.1	3.5

[a]500 ppm = 1,737 mg/m^3, 1,500 ppm = 5,210 mg/m^3, 3,500 ppm = 12,158 mg/m^3.
[b]Significant dose-related trend—Cochran-Armitage trend test $p < 0.05$.
[c]Significantly higher than control incidence by Fisher's exact test.
[d]– = Reported as "no exposure effect" by Burek et al. (1984); data not given.
[e]Burek et al. (1984) reported that 93/92 male mice had glomerulonephropathy in the kidney in the control group; the incidence was corrected to 92/92.
[f]Calculated by EPA.

Source: Burek et al. (1984).

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4124 In females, an increasing trend was seen in the incidence of foci or areas of altered
4125 hepatocytes. Female rats in all exposed groups showed increased incidence of multinucleated
4126 hepatocytes in the centrilobular region, compared with controls, but there was no evidence of

increasing incidence or severity with increasing exposure level (Table 4-21). The foci and areas were apparent after 12 months and their number and size increased thereafter, but incidences for neoplastic nodules in the liver or hepatocellular carcinomas were not increased in any exposure group. A statistically significant increased incidence of salivary gland sarcomas was reported for male rats exposed to 3,500 ppm. Burek et al. (1984) considered this finding unusual and inconsistent with other existing data because the primary target organ for dichloromethane seems to be the liver. Incidences of rats with benign mammary gland tumors were not statistically significantly higher in exposed male or female groups compared with controls, and exposed male and female groups showed no significantly increased incidences for malignant mammary gland tumors. The average number of benign mammary tumors per tumor-bearing rat increased with increasing exposure level. In females, the values were 2.1, 2.7, 3.1, and 3.5 in the control through 3,500 ppm groups, respectively; males showed a similar response with increasing exposure level, albeit to a lesser extent (Table 4-21). Burek et al. (1984) concluded that the significance of this benign mammary tumor response (i.e., increase in number of tumors per tumor-bearing rat) was unknown but speculated that the predisposition of this strain of rats (historical control incidences of female with benign mammary tumors normally exceeded 80%) plus the high exposure to dichloromethane may have resulted in the response.

4.2.2.2.5. ***Chronic inhalation exposure in Sprague-Dawley rats (Nitschke et al., 1988a).***

Nitschke et al. (1988a) examined the toxicity and carcinogenicity of lower concentrations of dichloromethane in Sprague-Dawley rats. Groups of 90 male and 90 female rats were exposed to 0, 50, 200, or 500 ppm dichloromethane (>99.5% pure) 6 hours/day, 5 days/week for 2 years. Interim sacrifices were conducted at 6, 12, 15, and 18 months (five rats/sex/interval). An additional group of 30 female rats was exposed to 500 ppm for 12 months and then exposed to room air for up to an additional 12 months, and another group of 30 female rats was exposed to room air for the first 12 months, followed by exposure to 500 ppm for the last 12 months of the study. These latter groups were included to examine temporal relationships between exposure and potential carcinogenic response. All groups of rats were examined daily for signs of toxicity and all rats were examined for palpable masses prior to the initial exposure and at monthly intervals after the first 12 months. BW was checked twice a month for the first 3 months and monthly thereafter. Blood samples were collected at interim sacrifices and analyzed for total bilirubin, cholesterol, triglycerides, potassium, estradiol, follicle-stimulating hormone, and luteinizing hormone levels. In addition, COHb was determined at multiple times in blood collected from the tail vein. DNA synthesis (incorporation of ^{3}H-thymidine as a measure of cellular proliferation) was measured in the liver of separate groups of female rats after exposure to the various concentrations for 6 and 12 months (four females/exposure group per interval). All rats were subjected to a complete necropsy, and sections from most tissues were processed for microscopic examination.

Exposure to dichloromethane at any of the exposure levels did not significantly alter mortality rates, BWs, organ weights, clinical chemistry values, or plasma hormone levels (Nitschke et al., 1988a). Blood COHb was elevated in a dose-related manner but not in an exposure duration-related fashion, suggesting lack of accumulation with repeated exposures. For example, mean (± SD) values for percentage COHb were 2.2 (± 1.3), 6.5 (± 1.1), 12.5 (± 0.8), and 13.7 (± 0.6) for male rats in the control through 500 ppm groups, respectively, at the terminal sacrifice and were similarly affected at the 6-month and 12-month intervals (e.g., respective values for males were 0.3(± 0.7), 2.8 (± 0.3), 9.6 (± 1.2), and 12.7 (± 1.6) at the 12-month sacrifice).

The results of the thymidine incorporation experiment revealed no detectable alteration in the rate of liver DNA synthesis in the exposed groups compared with controls. Statistically significantly increased incidences of nonneoplastic liver lesions (hepatic vacuolation and multinucleated hepatocytes) occurred only in females in the 500 ppm group (Table 4-22). Male rat incidence for hepatocyte vacuolation was elevated at 500 ppm but not to a statistically significant degree. In the group of female rats exposed for only 12 months to 500 ppm, significantly increased incidences of nonneoplastic lesions, compared with controls, were restricted to liver cytoplasmic vacuolization (16/25 = 64%) and multinucleated hepatocytes (9/25= 36%) in rats exposed during the first 12 months of the study; rats exposed only during the last 12 months of the study showed no elevated incidences of the liver lesions.

Table 4-22. Incidences of selected nonneoplastic histologic changes in male and female Sprague-Dawley rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm) [a]						
Lesion, by sex	Controls 0	50	200	500	Trend *p*-value[b]	Late 500[c]	Early 500[c]
Males—n per group	70	70	70	70		NA[d]	NA
Number (%) with							
Hepatic vacuolation (fatty change)	22 (31)	–[e]	–	28 (40)			
Multinucleated hepatocytes	–	–	–	–			
Females—n per group	70	70	70	70		25	25
Number (%) with							
Hepatic vacuolation (fatty change)	41 (59)	42 (60)	41 (59)	53 (76)[f]	0.01	15 (60)	16 (64)[f]
Multinucleated hepatocytes	8 (11)	6 (9)	12 (17)	27 (39)[f]	<0.0001	3 (12)	9 (36)[f]

[a]50 ppm = 174 mg/m^3, 200 ppm = 695 mg/m^3, 500 ppm = 1,737 mg/m^3.
[b]Cochran-Armitage trend test.
[c]Late 500 = no exposure for first 12 months followed by 500 ppm for last 12 months; early 500 = 500 ppm for first 12 months followed by no exposure for last 12 months.
[d]NA= there were no male rats in these exposure groups.
[e]– = Incidences not reported.
[f]Significantly ($p \leq 0.05$) higher than control incidence by Fisher's exact test (Nitschke et al., 1988a).

Source: Nitschke et al. (1988a).

A few fibrosarcomas or undifferentiated sarcomas in the mammary gland were seen in the exposed rats, but these incidences were not statistically significant (Table 4-23). Significantly increased incidences of rats with neoplastic lesions were restricted to benign mammary tumors in female rats exposed for 2 years to 200 ppm compared with controls (61/69 = 88%) (Table 4-23). However, significantly elevated incidences of this tumor type were not observed in 500 ppm females, and the 200 ppm incidence was within the range for historical control values for benign mammary tumors in female Sprague-Dawley rats (79–82%) from two other chronic toxicity/carcinogenicity studies from the same laboratory. A slight, but statistically significant, increase in the number of palpable masses in subcutaneous or mammary regions (at 23 months) per tumor-bearing rat was observed only in the 500 ppm female group. The numbers of benign mammary tumors per tumor-bearing rat were slightly elevated in the exposed groups compared with control groups, but no statistical analysis of this variable was performed. In female rats exposed to 500 ppm (during the first or second 12 months of the study), slight but statistically significant elevations were found in the number of palpable masses in subcutaneous or mammary regions per tumor-bearing rat; the numbers of benign mammary tumors per tumor-bearing rat were slightly elevated compared with those of controls, but statistical analysis of this variable was not performed.

A statistically significant increased incidence of brain or CNS tumors was not observed, but six astrocytoma or glioma (mixed glial cell) tumors were seen in the exposed groups (4 in males, 2 in females). The authors concluded that there was no distinct exposure-related malignant carcinogenic response in male or female Sprague-Dawley rats exposed (6 hours/day, 5 days/week) to up to 500 ppm dichloromethane for 2 years (Nitschke et al., 1988a).

Table 4-23. Incidences of selected neoplastic histologic changes in male and female Sprague-Dawley rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm) [a]					
	Controls				Late	Early
Lesion, by sex	0	50	200	500	500[b]	500[b]
Males—n per group	70	70	70	70	0	0
Number (%)[c] with						
Liver tumors	0 (0)	0 (0)	0 (0)	0 (0)		
Lung tumors	0 (0)	0 (0)	0 (0)	0 (0)		
Mammary gland tumors						
Adenocarcinoma or carcinoma	0 (0)	0 (0)	0 (0)	0 (0)		
Fibroadenoma	2 (4)	0 (0)	2 (3)	2 (3)		
Fibroma	6 (11)	1 (6)	6 (11)	10 (16)		
Fibrosarcoma	0 (0)	1 (6)	1 (6)	0 (0)		
Undifferentiated sarcoma	0 (0)	2 (4)	0 (0)	0 (0)		
Fibroma, fibrosarcoma, or undifferentiated sarcoma[d]	6 (11)	4 (6)	7 (12)	10 (16)		
Brain tumors						
Astrocytoma or glial cell	0 (0)	1 (1)	2 (3)	1 (1)		
Granular cell	0 (0)	0 (0)	0 (0)	1 (1)		
Females—n per group	70	70	70	70	25	25
Number (%)[c] with						
Liver tumors						
Neoplastic nodule(s)	4 (6)	4 (6)	3 (4)	4 (6)	0 (0)	1 (4)
Hepatocellular carcinoma	1 (1)	0 (0)	2 (3)	1 (1)	0 (0)	0 (0)
Lung tumors	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Mammary gland tumors						
Adenocarcinoma or carcinoma	6 (9)	5 (7)	4 (6)	4 (6)	3 (12)	2 (8)
Adenoma	1 (1)	1 (1)	2 (3)	1 (1)	2 (8)	0 (0)
Fibroadenoma	51 (74)	57 (83)	60 (87)	55 (80)	22 (88)	23 (92)
Fibroma	0 (0)	1 (1)	0 (0)	1 (1)	1 (4)	1 (1)
Fibrosarcoma	1 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Number with palpable masses in subcutaneous or mammary region	55 (78)	56 (81)	60 (87)	59 (86)	22 (88)	23 (92)
Number of palpable masses in subcutaneous or mammary region per tumor-bearing rat	1.8	2.1	2.0	2.2[e]	2.3[e]	2.7[e]
Number with benign tumors	52 (75)	58 (84)	61[f] (88)	55 (80)	23 (92)	23 (92)
Number of benign tumors per tumor-bearing rat	2.0	2.3	2.2	2.7	2.2	2.6
Brain tumors						
Astrocytoma or glial cell	0 (0)	0 (0)	0 (0)	2 (3)	0 (0)	0 (0)
Granular cell	1 (1)	0 (0)	0 (0)	1 (1)	0 (0)	0 (0)

[a]50 ppm = 174 mg/m^3, 200 ppm = 695 mg/m^3, 500 ppm = 1,737 mg/m^3

[b]Late 500 = no exposure for first 12 months followed by 500 ppm for last 12 months; early 500 = 500 ppm for first 12 months followed by no exposure for last 12 months. No males were included in these exposure groups.

[c]Percentages were based on the number of tissues examined microscopically per group. In males, 69 lungs were examined microscopically in the 50 ppm groups, and only 57, 65, 59, and 64 mammary glands were examined in the control, 50, 200, and 500 ppm groups, respectively. In females, 69 mammary glands were examined microscopically in the control, 50, 200, and 500 ppm groups.

[d]EPA summed across these three tumors, assuming no overlap.

[e]Significantly ($p \leq 0.05$) higher than control by Haseman's test (Nitschke et al., 1988a).

[f]Significantly ($p \leq 0.05$) higher than control incidence by Fisher's exact test (Nitschke et al., 1988a).

Source: Nitschke et al. (1988a).

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4.2.2.2.6. *Chronic inhalation exposure in Sprague-Dawley rats (Maltoni et al., 1988).* Maltoni et al. (1988) conducted an inhalation exposure study in Sprague-Dawley rats. Two groups of female rats (54–60/dose) were exposed to 0 or 100 ppm dichloromethane for 104 weeks. The exposure period was 4 hours/day, 4 days/week for 7 weeks and then 7 hours/day, 5 days/week for 97 weeks. Endpoints monitored included clinical signs, BW, and full necropsy at sacrifice (when spontaneous death occurred). For each animal sacrificed, histopathologic examinations were performed on the following organs: brain and cerebellum, zymbal glands, interscapular brown fat, salivary glands, tongue, thymus and mediastinal lymph nodes, lungs, liver, kidneys, adrenals, spleen, pancreas, esophagus, stomach, intestine, bladder, uterus, gonads, and any other organs with gross lesions.

There was no evidence of increased mortality in the exposed group, and there was no effect on BW (Maltoni et al., 1988). Little information was provided regarding nonneoplastic effects, precluding assignment of NOAELs and LOAELs for possible nonneoplastic effects in this study. Dichloromethane exposure was not related to the percentage of rats with benign tumors and malignant tumors, malignant tumors, or the number of total malignant tumors per 100 animals. The percentage of rats with benign mammary tumors was 40.0% in controls and 64.8% in the exposed group, and the percentage of malignant mammary tumors was 3.3 and 5.5% in controls and exposed, respectively. Neither of these differences was statistically significant.

4.3. REPRODUCTIVE/DEVELOPMENTAL STUDIES—ORAL AND INHALATION

Reproductive and development studies of dichloromethane exposure are summarized in Table 4-24 and described in detail below. No effects on reproductive performance were observed in a 90-day gavage study in Charles River CD rats with doses up to 225 mg/kg-day (General Electric Co., 1976) or in a two-generation reproductive toxicity study with F344 rats exposed to concentrations up to 1,500 ppm for 14 or 17 weeks before mating of the F0 and F1 generations, respectively, as well as during the F1 gestational period (GDs 0–21) (Nitschke et al., 1988b). Reproductive parameters (e.g., number of litters, implants/litter, live fetuses/litter, percent dead/litter, percent resorbed/litter, or fertility index[4]) were also examined in a study in male Swiss-Webster mice administered dichloromethane (250 or 500 mg/kg) by subcutaneous injection three times/week for 4 weeks, and in a similar study involving inhalation exposure to 0, 100, 150, or 200 ppm dichloromethane; no statistically significant effects were seen in either protocol, although some evidence of a decrease in fertility index was seen in the 150 and 200 ppm groups (Raje et al., 1988).

[4]Fertility index defined as number of females impregnated divided by total number of females mated times 100.

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Table 4-24. Summary of studies of reproductive and developmental effects of dichloromethane exposure in animals

Species and n	Exposure dose	Exposure period	Results	Reference
		Oral and Gavage		
Charles River rats (males and females), 10 per sex per dose group	0, 25, 75, 225 mg/kg (gavage)	90 days before mating (10 days between last exposure and mating period)	No effects on fertility index, number of pups per litter, pup survival, or F1 BW, hematology, and clinical chemistry tests (up to 90 days of age)	General Electric Co. (1976)
Swiss-Webster mice (males), 20 per group	0, 250, 500 mg/kg (subcutaneous injection), 3× per week	4 weeks prior to mating (1 week between last exposure and mating period)	No effects on fertility index, number of litters, implants per litter, live fetuses per litter, resorption rate; no testicular effects	Raje et al. (1988)
F344 rats (females), 17–21 per dose group	0, 337.5, 450 mg/kg-day (gavage)	GDs 6–19	Decreased maternal weight gain; no effect on resorption rate, number of live litters, implants, live pups, or pup weight	Narotsky and Kavlock (1995)
		Inhalation		
F344 rats (males and females, two generation), 30 per sex per dose group (F0 and F1)	0, 100, 500, 1,500 ppm, 6 hours/day	14 weeks prior to mating (F0), GDs 0–21, and 17 weeks prior to mating, beginning PND 4, (F1)	No effect on fertility index, litter size, neonatal survival, growth rates, or histopathologic lesions	Nitschke et al. (1988b)
Swiss-Webster mice (males), 20 per group	0, 100, 150, 200 ppm, 2 hours/day	6 weeks, prior to mating (2 days between last exposure and mating period)	Fertility index decreased in 150 and 200 ppm group (statistical significance depends on test used); no effects on number of litters, implants per litter, live fetuses per litter, resorption rate; no testicular effects.	Raje et al. (1988)
Long-Evans rats (female), 16–21 per dose group	0, 4,500 ppm	12–14 days before mating and/or GDs 1–17	Gestational exposure resulted in increased absolute and relative maternal liver weight, decreased fetal BW	Hardin and Manson (1980)
Long-Evans rats (female), 16–21 per dose group	0, 4,500 ppm	12–14 days before mating and/or GDs 1–17	Altered rate of behavioral habituation to novel environment (at 4 days of age). No effect on crawling (at 10 days), movement in photocell cage (15 days), use of running wheel (45–108 days), and shock avoidance (4 months).	Bornschein et al. (1980)

Table 4-24. Summary of studies of reproductive and developmental effects of dichloromethane exposure in animals

Species and n	Exposure dose	Exposure period	Results	Reference
Swiss-Webster mice (females), 30–40 per group	0, 1,250 ppm, 7 hours/day	GDs 6–15	Increased incidence of extra center of ossification in sternum, increased (~10%) maternal blood COHb, increased maternal weight, increased maternal absolute liver weight	Schwetz et al. (1975)
Sprague-Dawley rats (females), 20–35 per group	0, 1,250 ppm, 7 hours/day	GDs 6–15	Decreased incidence of lumbar ribs or spurs, increased incidence of delayed ossification of sternebrae, increased (~10%) maternal blood COHb, increased maternal absolute liver weight	Schwetz et al. (1975)

Following exposure of pregnant F344 rats to gavage doses of up to 450 mg/kg-day on GDs 6–19, maternal weight gain was decreased, but no effects were found on the number of resorption sites, pup survivability, or pup weights at postnatal days (PNDs) 1 or 6 (Narotsky and Kavlock, 1995). The developmental effects following exposure of Long-Evans rats to 4,500 ppm for 14 days prior to mating and during gestation (or during gestation alone) were decreased offspring weight at birth and changed behavioral habituation of the offspring to novel environments (Bornschein et al., 1980; Hardin and Manson, 1980) (see section 4.3.2 for more details). In standard developmental toxicity studies involving exposure to 1,250 ppm on GDs 6–15, no adverse effects on fetal development were found in Swiss-Webster mice or Sprague-Dawley rats, but the incidence of minor skeletal variants (e.g., delayed ossification of sternebrae) was increased. (Schwetz et al., 1975) (see section 4.3.2).

4.3.1. Reproductive Toxicity Studies

4.3.1.1. *Oral (Gavage) Studies*

In a study sponsored by the General Electric Co. (1976), Charles River CD rats (10/sex/dose level) were administered 0, 25, 75, or 225 mg/kg-day dichloromethane by gavage in water for 90 days. The test material was dichloromethane (of unspecified purity) purchased from Dow Chemical Company. At approximately 100 days of age, the rats were mated 1 to 1 to produce the F1 generation. F1 rats (15/sex/dose level) received the same treatment as F0 for 90 days, at which time they were sacrificed and necropsied. Comprehensive sets of 24 tissues from 10 male and 10 female F1 rats from the control and 225 mg/kg-day groups were examined microscopically after embedding, sectioning, and staining. F1 rats were monitored for clinical signs, BW effects, and food consumption. Reproductive parameters examined were fertility index, number of pups per litter, and pup survival. F1 rats also underwent hematology and clinical chemistry tests and urinalysis at 1, 2, and 3 months of the study and ophthalmoscopic examination at 3 months. There were no significant compound-related alterations in any of the endpoints monitored.

Raje et al. (1988) administered dichloromethane (250 or 500 mg/kg) by subcutaneous injection three times per week for 4 weeks to male Swiss-Webster mice (20/group). Mating with unexposed females started 1 week after the last exposure and continued for 2 weeks. After the mating period, the males were sacrificed, and the testes were examined microscopically. On GD 17, the females were sacrificed and the uterine horns examined for live, dead, or resorbed fetuses. The authors reported that exposure to dichloromethane had no statistically significant effects on number of litters, implants/litter, live fetuses/litter, percent dead/litter, percent resorbed/litter, or fertility index. Examination of the testes showed no significant alterations compared with controls.

4.3.1.2. *Inhalation Studies*

Nitschke et al. (1988b) conducted a two-generation reproductive toxicity study in rats. Groups of F344 rats (30/sex/dose level) were exposed by inhalation in whole-body chambers to 0, 100, 500, or 1,500 ppm dichloromethane (99.86% pure) 6 hours/day, 5 days/week for 14 weeks and then mated to produce the F1 generation. Exposure of dams continued after mating on GDs 0–21 but was interrupted until PND 4. After weaning 30 randomly selected F1 pups/sex/dose level were exposed as the parental generation for 17 weeks and subsequently mated to produce the F2 generation. The results showed no statistically significant exposure-related changes in reproductive performance indices (fertility, litter size), neonatal survival, growth rates, or histopathologic lesions in F1 (Table 4-25) or F2 weanlings sacrificed at time of weaning. According to the authors none of the values in Table 4-25 was significantly different from control values ($\alpha = 0.05$).

Table 4-25. Reproductive outcomes in F344 rats exposed to dichloromethane by inhalation for 14 weeks prior to mating and from GDs 0–21

	Exposure (ppm)[a]			
	0	**100**	**500**	**1,500**
Fertility index[b]	77%	77%	63%	87%
Gestation index[c]	100%	100%	100%	100%
Gestation survival index[d]	99.6%	100%	100%	96.6%
4-day survival index[e]	91.0%	95.2%	98.5%	98.6%
28-day survival index[f]	99.4%	99.4%	100%	99.5%
Sex ratio on day 1 (M:F)	48:52	50:50	50:50	52:48
Litter size				
Day 0	11 ± 2	10 ± 2	10 ± 3	11 ± 2
Day 28	7 ± 2	7 ± 2	7 ± 2	8 ± 2
Pup BWs, g				
Day 1	5.2 ± 0.4	5.3 ± 0.5	5.3 ± 0.4	5.2 ± 0.4
Day 4	7.4 ± 0.7	7.5 ± 1.1	7.7 ± 0.7	7.3 ± 0.7
Day 28, male	44.6 ± 5.8	45.9 ± 5.0	47.0 ± 5.4	45.0 ± 5.9
Day 28, female	43.2 ± 4.3	43.8 ± 4.5	44.4 ± 5.7	43.0 ± 4.8

[a]100 ppm = 347 mg/m^3, 500 ppm = 1,737 mg/m^3 , 1,500 ppm = 5,210 mg/m^3,
[b]Number of females delivering a litter expressed as a percentage of females placed with a male.
[c]Number of females delivering a live litter expressed as a percentage of the number of females delivering a litter.
[d]Percentage of newborn pups that were alive at birth.
[e]Percentage of pups surviving to day 4.
[f]Percentage of pups alive on day 4 and surviving to day 28.

Source: Nitschke et al. (1988b).

Raje et al. (1988) exposed groups of male Swiss-Webster mice (20/group) to 0, 100, 150, or 200 ppm dichloromethane (HPLC grade, JT Baker Chemical Co.) in inhalation chambers for 2 hours/day, 5 days/week for 6 weeks. Mating with unexposed females started 2 days after the last exposure. As in the subcutaneous injection protocol described in the previous section, after the 2-week mating period, the males were sacrificed and the females were sacrificed on GD 17. Exposure of the male mice to dichloromethane had no statistically significant effects on number of litters, implants/litter, live fetuses/litter, percent dead/litter, or percent resorbed/litter, and no significant alterations in the testes were noted. The fertility index was 95%, 95%, 80%, and 80% in the control, 100, 150 and 200 ppm groups, respectively. This decrease was not statistically significant as reported by the authors. Details of the statistical analyses were not provided. The overall Chi-square *p*-value was 0.27. Using a Cochran-Armitage exact trend test on these data, EPA calculated a one-sided *p*-value of 0.059. Individual *p*-values for the comparison of each group with the control group were 0.97, 0.17, and 0.17 for the 100, 150 and 200 ppm groups, respectively. The results for the combined 150 and 200 ppm groups were statistically different from the combined controls and 100 ppm group (Fisher's exact test, one-sided *p*-value = 0.048), suggesting a NOAEL of 100 ppm and LOAEL of 150 ppm.

4.3.2. Developmental Toxicity Studies

The metabolism of dichloromethane into CO by CYP2E1 raises concerns pertaining to developmental neurotoxicity. Gestational exposure to CO results in developmental toxicity and there are reports indicating that exposures as low as 75 ppm CO can result in significant neurological effects in offspring (Giustino et al., 1999). Neurobehavioral deficits in offspring include impaired avoidance behavior (De Salvia et al., 1995) and memory (Giustino et al., 1999). Neurochemical changes**,** such as abnormal dopaminergic function (Cagiano et al., 1998) and disruption of neuronal proliferation (Fechter, 1987)**,** have also been observed. Oral and inhalation dichloromethane exposure studies have demonstrated increased blood CO levels (see section 3.3). In addition, increased blood CO levels were seen in rat fetuses exposed through maternal inhalation to 500 ppm dichloromethane on GD 21 (Anders and Sunram, 1982), and placental transfer of dichloromethane also occurs (Withey and Karpinski, 1985; Anders and Sunram, 1982)

4.3.2.1. *Oral (Gavage) Studies and Culture Studies*

Narotsky and Kavlock (1995) evaluated developmental effects of dichloromethane (99.9% pure) in F344 rats (17–21/dose group) treated with 0, 337.5, or 450 mg/kg-day dichloromethane by gavage in corn oil on GDs 6–19. Dams were weighed on GDs 6, 8, 10, 13, 16, and 20 and allowed to deliver naturally. They were sacrificed on PND 6 to count uterine implantation sites. Pups were grossly examined for developmental abnormalities and weighed on PNDs 1, 3, and 6. Dead pups or pups with no gross abnormalities were sacrificed and

examined for soft tissue abnormalities. Maternal weight gain during pregnancy was significantly reduced in high-dose dams (by 33%, as estimated from Figure 5 of the paper); this group also exhibited rales and nasal congestion. Treatment with dichloromethane did not induce resorptions or alter the number of live litters on PND 1 or 6, the number of implants, the number of live pups on PND 1 or 6, or pup weight per litter. No gross or soft tissue abnormalities were observed.

Rat embryos in culture medium were exposed to 0, 3.46, 6.54, 9.79, or 11.88 μmol/mL dichloromethane for 40 hours. At the end of the exposure, embryos were observed for development of yolk sac vasculature, crown-rump length, total embryonic protein content, and number of somite pairs. A concentration of dichloromethane of 6.54 μmol/mL of culture medium resulted in decreased crown-rump length, decreased somite number, and decreased amount of protein per embryo, whereas no effects were seen at 3.46 μmol/mL (Brown-Woodman et al., 1998). A time-course experiment conducted with a concentration of dichloromethane of 9.22 μmol/mL showed that marked differences in growth and development from controls were not significant until about 8 hours of culture. Brown-Woodman et al. (1998) noted that the concentrations that caused embryotoxicity in this study were much higher than those found in individuals studied under controlled exposure conditions and comparable to those found in postmortem blood after fatal inhalation.

4.3.2.2. *Inhalation Studies*

Schwetz et al. (1975) exposed pregnant Swiss-Webster mice (30–40/group) and Sprague-Dawley rats (20–35/group) by inhalation in whole-body chambers to 0 or 1,250 ppm dichloromethane (97.86% pure) 7 hours/day on GDs 6–15. Maternal BWs were recorded on GDs 6, 10, and 16 and on the day of sacrifice (GD 18 for mice, GD 21 for rats). At sacrifice, uterine horns were excised and examined for fetal position and number of live, dead, or absorbed fetuses. Fetuses were observed for gross, soft tissue, and skeletal abnormalities. The only effects seen on developing fetuses were changes in the incidence of minor skeletal variants. In rats, the incidence of lumbar ribs or spurs was significantly decreased compared with controls, whereas the incidence of delayed ossification of sternebrae was significantly greater than in controls. In mice, a significant number of litters contained pups with a single extra center of ossification in the sternum. Exposure to dichloromethane produced significantly elevated blood COHb content in dams of both species (approximately 9–10% after 10 exposures versus 1–2% in controls). BWs in exposed mouse dams were significantly increased (11–15%) compared with those in controls but were not affected in exposed rat dams. Mean absolute liver weights of exposed dams of both species were significantly elevated compared with controls, but mean relative liver weights were not affected. The results indicate that 1,250 ppm was a LOAEL for minimal maternal effects (increased COHb and increased absolute liver weight) and a LOAEL for adverse effects on the fetuses.

Hardin and Manson (1980) conducted a study in female Long-Evans rats to determine whether exposure before and during gestation is more detrimental to reproductive outcome than exposure either before or during gestation alone. Four groups of 16–21 rats were formed in which the rats were exposed by inhalation in whole-body chambers to 4,500 ppm dichloromethane (technical grade, >97% pure) 6 hours/day for 12–14 days before breeding and/or on GDs 1–17 or were exposed to filtered air. Maternal BWs were measured every 4 days. Dams were euthanized on GD 21 and livers and uteri removed. Livers were weighed, and uterine horns were examined for fetal position and number of live, dead, or absorbed fetuses. Fetuses were observed for gross, soft-tissue, and skeletal abnormalities. Exposure during gestation (with or without pre-gestation exposure) significantly increased maternal liver weight (absolute and relative) by about 10–12 and 9–12%, respectively, and decreased fetal BW by about 9–10% relative to those exposed to filtered air during gestation. None of the groups showed significant alterations in the incidence of gross, external, skeletal, or soft-tissue anomalies. Using the same study design and exposure level, Bornschein et al. (1980) observed behavioral activities at various ages. Assessed activities included head movement/pivoting when placed in a novel environment (4 days of age), limited crawling (10 days), movement in a photocell cage (15 days), use of running wheel (45–108 days), and shock avoidance (4 months). Exposure during gestation (with or without pre-gestation exposure) caused altered rates of behavioral habituation to novel environments in the pups tested as early as 10 days of age that were still present at 150 days of age. Growth, food and water consumption, wheel running activity, and avoidance learning were not significantly affected by exposure to dichloromethane. The results indicate that 4,500 ppm was a LOAEL for maternal effects (10% increased absolute and relative liver weight) and for effects on the fetuses (10% decreased fetal BW and altered behavioral habituation to novel environments).

In a study of early-life (including gestational) exposures, Maltoni et al. (1988) exposed 54 pregnant Sprague-Dawley rats to 100 ppm dichloromethane via inhalation 4 hours/day, 5 days/week for 7 weeks, followed by 7 hours/day, 5 days/week for 97 weeks. Exposure apparently started on GD 12. Groups of 60 male and 69 female newborns continued to be exposed after birth to 60 ppm dichloromethane 4 hours/day, 5 days/week for 7 weeks, followed by exposure 7 hours/day, 5 days/week for 97 weeks. Additional groups of 60 male and 70 female newborn were exposed after birth to 60 ppm dichloromethane 4 hours/day, 5 days/week for 7 weeks and then for 7 hours/day, 5 days/week for 8 weeks. BWs were measured every 2 weeks during exposure and every 8 weeks thereafter. At the end of exposure, animals were sacrificed and histologic examinations were performed on 20 tissue types.

Early life exposures of Sprague-Dawley rats to dichloromethane (Maltoni et al., 1988) did not affect mortality or BW in any group. Also, there was no significant effect of exposure to dichloromethane on the percentage of animals with benign and malignant tumors and malignant tumors, the number of malignant tumors per 100 animals, or the percentage of animals with

benign mammary tumors, malignant mammary tumors, leukemias, pheochromocytomas, and pheochromoblastomas. The results provide no evidence that gestational exposure to 100 ppm dichloromethane during early life stages of development increases the susceptibility of Sprague-Dawley rats to the potential carcinogenicity of dichloromethane, but further conclusions from these results are precluded because the study included only one exposure level that was below the maximum tolerated dose for adult Sprague-Dawley rats. Experiments comparing cancer responses from early-life exposures with adult exposures are not available for F344 rats or $B6C3F_1$ mice, the strains of animals in which carcinogenic responses to dichloromethane have been observed.

In summary, the potential for gestational exposure to CO, resulting from maternal dichloromethane exposure via oral and inhalation routes, raises concerns regarding neurodevelopmental effects. In addition, dichloromethane transfer across the placenta has also been seen in inhalation exposure studies in rats (Withey and Karpinski, 1985; Anders and Sunram, 1982). Although few developmental effects were observed at high exposures of dichloromethane (Bornschein et al., 1980; Schwetz et al., 1975), there are no studies that have thoroughly evaluated neurobehavioral and neurochemical changes resulting from gestational dichloromethane exposure. The available data identify changes of behavior habituation at 4,500 ppm (Bornschein et al., 1980) and increases in COHb at 1,250 ppm (Schwetz et al., 1975). The behavioral changes observed at 4,500 ppm indicate developmental neurotoxic effects. No other neurological endpoints have been evaluated in the available developmental studies of dichloromethane, but increases in blood COHb strongly suggest that dichloromethane is being metabolized to CO. Gestational exposure to CO can result in significant neurological effects in offspring, including neurobehavioral deficits (De Salvia et al., 1995), memory effects (Giustino et al., 1999), and neurochemical changes (Cagiano et al., 1998; Fechter, 1987). As a result, it is unknown if developmental neurotoxicity could occur at lower exposures to dichloromethane.

4.4. OTHER DURATION- OR ENDPOINT-SPECIFIC STUDIES

4.4.1. Short-term (2-Week) Studies of General and Hepatic Effects in Animals

Two short-term (2-week) studies examined hepatic and renal effects of dichloromethane exposure in F344 rats (Berman et al., 1995) and CD-1 mice (Condie et al., 1983). Berman et al. (1995) administered dichloromethane by gavage in corn oil for up to 14 days to groups of eight female F344 rats at dose levels of 0, 34, 101, 337, or 1,012 mg/kg-day. Starting at day 4, deaths occurred in the 1,012 mg/kg-day exposure group, with seven of eight rats dying before the end of the 14-day exposure period. In the dose groups that did not experience this high mortality, incidences of increased necrotic hepatocytes were 0/8, 0/8, 0/8, and 3/8 for the 0, 34, 101, and 337 mg/kg-day groups, respectively. The increase in liver lesions was not accompanied by increases in serum activities of ALT or AST. Kidneys, spleen, and thymus were also histopathologically examined in this study, but none showed exposure-related lesions. The

results indicate that 101 mg/kg-day was a NOAEL and 337 mg/kg-day was a LOAEL for increased incidence of degenerative lesions in female rats exposed for 14 days. In a companion study with groups of eight female F344 rats that were given single doses of 0, 101, 337, 1,012, or 1,889 mg/kg-day, incidences of rats with increased necrotic hepatocytes were 1/8, 0/8, 8/8, 7/8, and 8/8, respectively (Berman et al., 1995).

Condie et al. (1983) detected exposure-related liver lesions in a 14-day gavage study in which dichloromethane in corn oil was administered to male CD-1 mice at dose levels of 0, 133, 333, or 665 mg/kg-day. Incidences of mice with minimal or slight cytoplasmic vacuolation were 1/16, 0/5, 3/5, and 4/5 for the control through high-dose groups, respectively. The kidneys were also examined histopathologically in this study but showed no exposure-related lesions. No other tissues were prepared for histologic examination. Blood urea nitrogen, serum creatinine, and serum ALT activities were not significantly altered by exposure. All dose levels significantly reduced to the same extent the active transport of p-aminohippurate into renal cortical slices in vitro, a measure of proximal tubule function. The results most clearly identify 133 mg/kg-day as a NOAEL and 333 mg/kg-day as a LOAEL for increased incidence of hepatocyte vacuolation in male mice.

4.4.2. Immunotoxicity Studies in Animals

Aranyi et al. (1986) studied the effects of acute inhalation exposures to 50 or 100 ppm dichloromethane on two measures of immune response (susceptibility to respiratory infection and mortality due to *Streptoccocus zooepidemicus* exposure and ability of pulmonary macrophages to clear infection with *Klebsiella pneumoniae*). Female CD1 mice that were 5–7 weeks of age at the start of the exposure portion of the experiment were used for both assays. Up to five replicate groups of about 30 mice were challenged with viable *S. zooepidemicus* during simultaneous exposure to dichloromethane or to filtered air. Deaths were recorded over a 14-day observation period. Clearance of ^{35}S-labeled *K. pneumoniae* by pulmonary macrophages was determined by measuring the ratio of the viable bacterial counts to the radioactive counts in each animal's lungs 3 hours after infection; 18 animals were used per dose group. A single 3-hour exposure to 100 ppm dichloromethane significantly increased the susceptibility to respiratory infection and greater mortality following exposure to *S. zooepidemicus* ($p \leq 0.01$). Twenty-six deaths occurred in 140 (18.6%) mice challenged during a 3-hour exposure to 100 ppm dichloromethane; in contrast, nine deaths occurred in 140 mice (6.4%) exposed to filtered air. The 3-hour exposure to 100 ppm dichloromethane was associated with a statistically significant ($p \leq 0.001$) 12% decrease in pulmonary bactericidal activity (91.6 and 79.6% of bacteria killed in controls and 100 ppm group, respectively). No difference was seen in either mortality rate or bactericidal activity in experiments using a single 3-hour exposure to 50 ppm or 3-hour exposures to 40 ppm dichloromethane repeated daily for 5 days compared with control animals exposed to filtered air. These results suggest that 3-hour exposure to 50 ppm

dichloromethane was a NOAEL and 100 ppm was a LOAEL for decreased immunological competence (immunosuppression) in CD-1 mice.

Aranyi et al. (1986) also conducted a similar set of experiments with 13 other chemicals (acetaldehyde, acrolein, propylene oxide, chloroform, methyl chloroform, carbon tetrachloride, allyl chloride, benzene, phenol, monochlorobenzene, benzyl chloride, perchloroethylene, and ethylene trichloride). Perchloroethylene and ethylene trichloride were the only chemicals in this group for which an increased mortality risk from streptococcal pneumonia was seen (mortality risk 15.0 and 31.4% in controls and 50 ppm exposure groups for perchloroethylene and 13.4 and 58.1% in controls and 50 ppm exposure groups for ethylene trichloride). Decreased bactericidal activity was also seen with acetaldehyde, acrolein, methyl chloroform, allyl chloride, benzene, benzyl chloride, perchloroethylene, and ethylene trichloride at one or more exposures. Results from several chemicals suggest that 5 days of exposure results in greater decrease in bactericidal activity (i.e., acetaldehyde, acrolein, and benzene), and others (e.g., perchloroethylene) suggest that 5 days of exposure does not result in greater suppression than a single exposure period.

There was considerable variation in both measures of immune response among the controls in the experiments (Aranyi et al., 1986). Among the controls in the experiments with the 13 chemicals other than dichloromethane, mortality in the streptococcal infectivity model ranged from 5.7–22.1%, with a mean of 12.7%.[5] Bactericidal activity in the klebsiella model among controls ranged from 67.9–94.7%, with a mean of 81.8%. The number of bacteria deposited in the lung in an inhalation bacterial infectivity model can show considerable variation, (i.e., between 750 to 1,500 viable streptococcus or klebsiella organisms, [Ehrlich, 1980]). Therefore, concurrent controls are particularly import due to the variation in preparation and aerosol administration of the bacteria in these assays.

Warbrick et al. (2003) evaluated immunocompetence in male and female Sprague-Dawley rats by measuring the immunoglobulin M (IgM) antibody responses following immunization with sheep red blood cells in addition to hematological parameters and histopathology of the spleen, thymus, lungs, and liver. Groups of rats (8/sex/dose level) were exposed to 0 or 5,000 ppm dichloromethane 6 hours/day, 5 days/week for 28 days. Rats injected with cyclophosphamide served as positive controls. Five days before sacrifice (day 23 of exposure) all rats were injected with sheep red blood cells. IgM levels in response to the sheep red blood cells were comparable between dichloromethane-exposed and air-exposed rats, indicating that dichloromethane did not produce immunosuppression in the animals under these exposure conditions. Cyclophosphamide-treated animals had significantly lower levels of IgM in the blood serum, indicating immunosuppression. Rats exposed to dichloromethane showed reduced response to sound, piloerection, and hunched posture during exposures. Neither BW gain nor the hematological parameters monitored were significantly affected by exposure to

[5]EPA did not include the duplicate assay of perchloroethylene in calculating this summary statistic. If this additional assay is included, the mortality risk ranges from 5.7–45.7%, with a mean of 15.0%.

dichloromethane. Relative and absolute liver weights were significantly increased in females, but not in males. Relative spleen weight was reduced in females, and no significant changes were seen in the weight of the thymus and lungs. Histopathology of the tissues examined was unremarkable. Exposure to 5,000 ppm dichloromethane did not affect antibody production to the challenge with sheep red blood cells.

In the 2-year drinking water study (Serota et al., 1986a, b) and 2-year inhalation study (Nitschke et al., 1988a), histopathologic analyses were conducted on the lymph nodes, thymus, and spleen among several other organs, and no significant changes were noted.

In summary, one study (Aranyi et al., 1986) demonstrated evidence of immunosuppression, including increased risk of streptococcal-pneumonia-related mortality and decreased clearance of klebsiella bacteria following a single dichloromethane exposure at 100 ppm for 3 hours in CD-1 mice. The streptococcal and klebsiella bacterial inhalation assays are models of respiratory infection that test for local immune effects associated with inhalation exposure rather than systemic immunosuppression. The NOAEL identified in this study was 50 ppm. In contrast, in a functional immune assay of systemic immunosuppression conducted in rats, Warbrick et al. (2003) did not observe changes in the antibody response to sheep red blood cells in a 28-day inhalation exposure to 5,000 ppm dichloromethane. Histopathologic analyses of immune system organs in chronic exposure studies for $B6C3F_1$ mice and F344 rats (Nitschke et al., 1988a; Serota et al., 1986a, b) revealed no changes from controls. However, no assays of functional immunity were included in these chronic studies. The limited database for dichloromethane does not suggest systemic immunosuppression, but the Aranyi et al. (1986) study provides evidence of route-specific local immunosuppression from acute exposure studies in CD1 mice. Due to the acute exposure duration used in Aryani et al. (1986), the immune effects of short-term or chronic exposure to dichloromethane are unclear.

4.4.3. Neurotoxicology Studies in Animals

Neurological evaluations in animals during and after exposure to dichloromethane have resulted in CNS depressant effects similar to other chlorinated solvents (e.g., trichloroethylene, perchloroethylene) and ethanol. Overall, there are decreased motor activity, impaired memory, and changes in responses to sensory stimuli. Neurobehavioral, neurophysiological, and neurochemical/ neuropathological studies have been used to characterize the effects of dichloromethane on the CNS. A brief overview of these types of studies is provided below, followed by a detailed description of individual studies.

Neurobehavioral studies with dichloromethane used protocols to measure changes in spontaneous motor activity, a functional observational battery (FOB) test (to evaluate gross neurobehavioral deficits), and a task developed to assess learning and memory. The FOB protocol includes various autonomic parameters, neuromuscular parameters, sensorimotor parameters, excitability measures, and activity. Learning and memory changes with

dichloromethane were studied by using a passive avoidance task. The oral and inhalation studies that examined neurobehavioral endpoints are summarized in Table 4-26.

Neurophysiological studies with dichloromethane exposure consisted of measuring evoked responses in response to sensory stimuli. In these studies, animals were implanted with electrodes over the brain region that responds to the particular stimuli. For example, an electrode would be implanted over the visual cortex in an animal presented with a visual stimulus. Once the stimulus is presented to the animal, an evoked response is elicited from the brain region and transmitted to the implanted electrode. During administration of a chemical, if there is a significant change in the magnitude, shape, and latency (among other measures) in the evoked response, then the chemical is considered to produce neurological effects. A summary of studies examining dichloromethane exposure and neurophysiological changes is shown in Table 4-27.

In neurochemical/neuropathological studies with dichloromethane, animals were first exposed to dichloromethane (via oral, inhalation, or injection), and then the brains were removed. Changes in excitatory neurotransmitters, such as glutamate and acetylcholine, and the inhibitory neurotransmitter, GABA, were measured. Additionally, dopamine and serotonin levels, which are associated with addiction and mood, were also measured. Other parameters that were measured included DNA/protein content and regional brain changes in the cerebellum and hippocampus. Measurement of neurochemical changes provides mechanistic information, and neurobehavioral and neurophysiological effects can be correlated to these results. Table 4-28 summarizes studies of neurochemical changes and dichloromethane.

Table 4-26. Studies of neurobehavioral changes from dichloromethane, by route of exposure and type of effect

Species	Exposure(s)	Duration	Neurobehavioral effect	Reference
		Oral and gavage exposure		
		Functional observational battery		
F344 rat, female	101, 337, 1,012, 1,889 mg/kg, gavage	Acute—evaluated 4 and 24 hours after dosing	FOB neuromuscular and sensorimotor parameters significantly different from controls at 1,012 and 1,889 mg/kg (337 mg/kg = NOAEL)	Moser et al. (1995)
F344 rat, female	34, 101, 337, 1,012 mg/kg-day, gavage	14 day—evaluated on days 4, 9, and 15.	All FOB parameters (except activity) significantly affected from day 4 at doses of 337 and 1,012 mg/kg-day	Moser et al. (1995)
		Inhalation exposure		
		Spontaneous activity		
NMRI mouse, male	400–2,500 ppm	1 hour	Initial increase in activity followed by a pronounced decrease at exposures 600 ppm and higher	Kjellstrand et al. (1985)
Rat, male	5,000 ppm	1 hour, every other day for 10 days	Decreased spontaneous locomotor activity	Heppel and Neal (1944)
Wistar rat, male	500 ppm	6 hours/day, 6 days	Increased preening frequency	Savolainen et al. (1977)
ICR mouse, female	5,000 ppm	Continuous, 7 days	Increased spontaneous activity in first few hours and then decreased activity	Weinstein et al. (1972)
Sprague-Dawley rat, male	1,000, 5,000 ppm	Continuous, 14 weeks	No neurobehavioral changes	Haun et al. (1971)
ICR mouse, female	1,000, 5,000 ppm	Continuous, 14 weeks	Incoordination, lethargy	Haun et al. (1971)
Beagle dog, female	1,000, 5,000 ppm	Continuous, 14 weeks	Incoordination, lethargy	Haun et al. (1971)
Rhesus monkey, female	1,000, 5,000 ppm	Continuous, 14 weeks	Incoordination, lethargy	Haun et al. (1971)
ICR mouse, female	25, 100 ppm	Continuous, 14 weeks	Increased spontaneous activity at 25 ppm	Thomas et al. (1972)
		Functional observational battery		
F344 rat, male and female	50, 200, 2,000 ppm	6 hours/day, 5 days/week, 13 weeks + 65 hours exposure free	No effects observed on FOB, grip strength	Mattsson et al. (1990)
		Learning and memory		
Swiss-Webster mouse, male	47,000 ppm	Approximately 20 seconds + 1 hour exposure free before training; retested at days 1, 2, and 4	Significant decrease in learning and recall ability	Alexeef and Kilgore (1983)

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Table 4-27. Studies of neurophysiological changes as measured by evoked potentials resulting from dichloromethane, by route of exposure

Species	Exposure(s)	Duration	SEPs[a] measured	Effect	Reference
			Intraperitoneal		
Long-Evans rat, male	57.5, 115, 230, 460 mg/kg, i.p.[a]	Acute; tested at 15 minutes, 1 hour, and 5 hours after dosing	FEP[a]	Significant changes in FEPs were noted in animals dosed 115 mg/kg and higher; FEP changes time and dose dependent	Herr and Boyes (1997)
			Inhalation Exposure		
F344 rat, male	5,000, 10,000, 15,000 ppm	Acute, 1 hour; tested during exposure	Electroencephalogram, BAER[a], CAEP[a], FEP, SEP	Significant changes in SEP, FEP, BAER, and CAEP responses at all exposures; slight recovery noted at 1 hour after exposure	Rebert et al. (1989)
F344 rat, male and female	50, 200, 2,000 ppm	Subchronic, 6 hour/day, 5 days/week, 13 weeks; tested 65 hours after last exposure	FEP, CAEP, BAER, SEP	No significant changes noted in any evoked potential measurements	Mattsson et al. (1990)

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4591 [a]SEP = somatosensory-evoked potential; FEP = flash-evoked potential; BAER = brainstem-auditory-evoked response; CAEP = cortical-auditory-evoked
4592 potential; i.p. = intraperitoneal.
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Table 4-28. Studies of neurochemical changes from dichloromethane, by route of exposure

Species and sex	Exposure	Duration	Regions	Effect[a]	Reference
			Oral exposure		
Sprague-Dawley rat, male	534 mg/kg	Acute, single dose; evaluated 2 hours after dosed	Hippocampus, medulla, midbrain, hypothalamus	↑ acetylcholine in hippocampus ↑ dopamine and serotonin in medulla ↓ norepinephrine in midbrain ↓ norepinephrine and serotonin in hypothalamus	Kanada et al. (1994)
			Inhalation exposure		
Wistar rat, male	1,000 ppm TWA (basal exposure of 100 ppm + 2,800 ppm, 1 hour peak exposures at hours 1 and 4)	6 hours/day, 5 days/week, 2 weeks	Cerebrum, cerebellum	↑ NADPH diaphorase, succinate dehydrogenase in cerebrum ↑ cerebral RNA ↓ succinate dehydrogenase in cerebellum	Savolainen et al. (1981)
Wistar rat, male	1,000 ppm TWA	6 hours/day, 5 days/week, 2 weeks + 7 days exposure free	Cerebrum, cerebellum	↓ succinate dehydrogenase in both regions	Savolainen et al. (1981)
Wistar rat, male	1,000 ppm	6 hours/day, 5 days/week, 2 weeks	Cerebrum, cerebellum	↑ acid proteinase ↓ succinate dehydrogenase in cerebellum	Savolainen et al. (1981)
Wistar rat, male	1,000 ppm	6 hours/day, 5 days/week, 2 weeks + 7 days exposure free	Cerebrum	↓ cerebral RNA	Savolainen et al. (1981)
Sprague-Dawley rat, male	70, 300, 1,000 ppm	6 hours/day, 3 days	Caudate nucleus—medial	↑ catecholamine levels (70 ppm) ↓ catecholamine levels (300 and 1,000 ppm) No effect on luteinizing hormone release	Fuxe et al. (1984)
Mongolian gerbil, male and female	210, 350 ppm	Continuous (24 hours/day), 3 months + 4 months exposure free	Hippocampus, cerebellum cerebral cortex	↓ DNA concentration per wet weight in hippocampus (210, 350 ppm) and cerebellar hemispheres (350 ppm) ↑ astroglial proteins in frontal and sensory motor cerebral cortex	Rosengren et al. (1986)
Mongolian gerbil, male and female	210 ppm	Continuous (24 hours/day), 3 months	Frontal cortex, cerebellum	↓ glutamate, GABA, phosphoethanolamine in frontal cortex ↑ glutamate, GABA in posterior cerebellar vermis	Briving et al. (1986)
Mongolian gerbil, male and female	210 ppm	Continuous (24 hours/day), 3 months + 4 months exposure free	Hippocampus, olfactory bulbs, cerebral cortex	↓ DNA concentration per wet weight in hippocampus only	Karlsson et al. (1987)

[a]↑ = increase; ↓ = decrease.

4.4.3.1. *Neurotoxicology Studies—Oral Exposures*

Three studies evaluated the neurotoxic potential of dichloromethane by either administering the solvent orally or by injection; two of these studies (Herr and Boyes, 1997; Kanada et al., 1994) only evaluated acute effects (2–5 hours) from single-dose exposures. Observed neurological effects included decreased spontaneous activity (Moser et al., 1995), changes in flash-evoked potential (FEP) measurements (Herr and Boyes, 1997), and changes in catecholamine levels in the brain (Kanada et al., 1994).

Moser et al. (1995) conducted neurobehavioral evaluations in female F344 rats following an acute or 14-day oral administration of dichloromethane. A FOB protocol was utilized to determine changes in autonomic parameters (lacrimation, salivation, pupil response, urination, defecation), neuromuscular parameters (gait, righting reflex, forelimb and hind-limb grip strength, landing foot splay), sensorimotor parameters (tail pinch, click response, touch response), excitability measures (handling reactivity, arousal, clonic, and/or tonic movements), and activity (rearing, motor activity). A baseline FOB was performed on all rats prior to initial dichloromethane administration. After dichloromethane administration, a FOB was conducted at selected time points followed by a motor activity test in a maze. In the acute study, rats were dosed with 0, 101, 337, 1,012, or 1,889 mg/kg dichloromethane. At 4 and 24 hours after the administered dose, rats were tested for the neurological parameters. Significant changes in the neuromuscular and sensorimotor parameters were observed and occurred mostly in rats administered with the highest dose. These significant changes were only observed at the 4-hour time point and not when measured at 24 hours. The NOAEL identified by the authors for this study was 337 mg/kg, based on no observable changes in the FOB. In the 14-day study, rats were administered 0, 34, 101, 337, or 1,012 mg/kg-day. FOB testing was conducted on days 4 and 9 (before the daily dose) and approximately 24 hours after the last (14th) dose. With the exception of the activity measurements, all other neurobehavioral parameters (neuromuscular, sensorimotor, autonomic, excitability) were significantly affected from the 4th day through the entire 14-day exposure cycle. The NOAEL identified for the 14-day study was 101 mg/kg-day, based on FOB changes associated with the dichloromethane exposure.

A single dose acute neurophysiology study by Herr and Boyes (1997) evaluated the effect of dichloromethane on FEPs in adult male Long-Evans rats. Rats were implanted with epidural electrodes over the visual cortex area. After placement in an enclosed rectangular mirror chamber, PEPs were stimulated with a 10 μsec flash. Baseline FEPs were collected and rats were injected intraperitoneally with 0 (corn oil, n = 16), 57.5 (n = 15), 115 (n = 15), 230 (n = 14), or 460 (n = 15) mg/kg dichloromethane. Animals were retested at 15 minutes, 1 hour, and 5 hours after injection. Amplitude decreases in the early FEP components were observed. The FEP amplitude changes were time and dose dependent with maximal effects at 15 minutes after dichloromethane dosage. All of the waveform amplitudes returned to control levels when measured at the 1-hour time point for all doses tested. Response latencies were still different

from controls when measured 5 hours after dosing, but the effect was less pronounced than at the 15-minute and 1-hour time points. In this study, 57.5 mg/kg did not produce any significant changes in the FEP measures as compared to control and was considered this study's NOAEL. The LOAEL was 115 mg/kg based on changes in the FEP amplitudes.

Kanada et al. (1994) examined the effect of dichloromethane on acetylcholine and catecholamines (dopamine, norepinephrine, serotonin) and their metabolites in the midbrain, hypothalamus, hippocampus, and medulla from male Sprague-Dawley rats (4–5/group) in a neurochemical/neuropathology study. The rats were sacrificed 2 hours after a single gavage dose of 0 or 534 mg/kg of undiluted dichloromethane. Administration of dichloromethane significantly increased the concentration of acetylcholine in the hippocampus by approximately 10% and increased dopamine and serotonin levels in the medulla by approximately 75%. Dichloromethane decreased norepinephrine levels in the midbrain and hypothalamus by 12–15%, and serotonin levels were decreased in the hypothalamus by approximately 30%. There was a trend toward decreased dopamine in the hypothalamus, but the variability between the animals was so high that the effect was not significant. (These values for the percent changes were estimated by EPA from the figures presented in the paper.) The authors speculated that increased acetylcholine release associated with exposure to dichloromethane and other solvents may originate from the nerve terminals.

4.4.3.2. *Neurotoxicology Studies—Inhalational Exposure*

The database pertaining to neurotoxic effects from inhalation exposure to dichloromethane is considerably larger than the oral exposure database. Acute (less than 1 day) and short-term (1–14 days) exposures resulted in an initial increase in spontaneous activity followed by a decrease for exposures between 500 and 2,500 ppm (Kjellstrand et al., 1985; Savolainen et al., 1977). Higher (5,000 ppm) acute and short-term exposures resulted in decreased spontaneous activity and lethargy (Weinstein et al., 1972; Heppel and Neal, 1944). Longer-term exposures (up to 14 weeks) produced decreased motor activity and lethargy in several animals at 1,000 and 5,000 ppm (Haun et al., 1971), and exposures at 25 ppm for 14 weeks produced significant increases in activity in mice, starting at week 9. CNS depression was evidenced by decreased responses in the auditory, visual, and somatosensory regions of the brain in a study of sensory-evoked potential effects in 12 adult male F344 rats exposed to 0, 5,000, 10,000, and 15,000 ppm for 1 hour periods (Rebert et al., 1989). Altered learning and memory abilities were demonstrated in young (3-, 5-, and 8-week-old) male Swiss-Webster mice exposed to 168 mg/L (~47,000 ppm) dichloromethane for approximately 20 seconds (until there was a loss of the righting reflex) (Alexeef and Kilgore, 1983).

4.4.3.2.1. ***Inhalational exposure—neurobehavioral studies.***

Spontaneous motor activity—acute and short-term studies

Heppel and Neal (1944) evaluated the neurological effects of 5,000 ppm dichloromethane in five male rats by measuring changes in spontaneous activity during and after exposure. The five rats were not randomly selected, since the investigators chose to pick out the most active animals in the litter. During the 1-hour testing runs, rats were placed in a rotating drum. Spontaneous activity was reported as the number of drum revolutions/hour. Twenty control test runs (1 run/day) were conducted prior to dichloromethane exposure runs. After the pre-exposure period, rats were exposed to 5,000 ppm dichloromethane every other day for 1 hour, and activity was measured in the same manner as in the control runs. Once dichloromethane exposure was stopped, the animals were allowed to recover for 30 minutes and a second 1-hour test run was performed to evaluate spontaneous activity during recovery. On nonexposure days, spontaneous activity was also measured in1-hour intervals to compare to the pre-exposure period. A total of five dichloromethane exposures, five postexposure, and five nonexposure trials were conducted over 10 days. Spontaneous activity significantly declined ($p < 0.01$, Fisher's t-test) during exposure to 5,000 ppm dichloromethane in comparison to nonexposure days. The average number of revolutions for all five rats over the test runs was 576 on nonexposure days and 59 revolutions during dichloromethane exposure.

Weinstein et al. (1972) continuously exposed female ICR mice to 5,000 ppm dichloromethane for up to 7 days. Clinical behavioral observations of the mice were made during dichloromethane exposure. Within the first few hours of exposure, spontaneous activity increased in comparison to control animals. After 24 hours of continuous exposure, there was a considerable decrease in spontaneous activity as noted by observation only. The mice also appeared to be very lethargic and had a hunched posture and a rough hair coat, which are all signs of CNS depressive effects in rodents. These effects became progressively worse until after 96 hours of exposure, where many mice resumed normal activity. After the 7-day exposure, mice were nearly as active as the control animals but had a rougher coat and were judged to be emaciated and dehydrated.

Male Wistar rats exposed to 500 ppm dichloromethane 6 hours/day for 6 days exhibited an increase in preening frequency and time 1 hour after the last exposure relative to controls (Savolainen et al., 1977). However, there were no significant changes in other types of spontaneous activity.

In the study by Kjellstrand et al. (1985), male NMRI mice were exposed to dichloromethane concentrations ranging from 400 to 2,500 ppm. At concentrations of 600 ppm and higher, exposures for 1 hour produced a biphasic pattern of activity characterized by an initial increase in activity (as high as 200% of preexposure motor activity at 2,200 ppm, as estimated from Figure 6 in Kjellstrand et al. [1985]) during exposure followed by a decreased that reached the lowest point 1–2 hours after the end of exposure (as low as 40% motor activity

at 2,200 ppm, in comparison to preexposure, as estimated from Figure 6 in Kjellstrand et al. [1985]). Motor activity returned to normal levels after the decreased activity observed 1–2 hours after exposure was stopped and indicated that the effect was reversible in this study design.

Spontaneous motor activity—subchronic (14 week) studies

Haun et al. (1971) reported results from studies in which female beagle dogs, female rhesus monkeys, male Sprague-Dawley rats, and female ICR mice were continuously exposed to 0, 1,000, or 5,000 ppm dichloromethane for up to 14 weeks in whole-body exposure chambers. Gross and histopathologic examinations were made on animals that died or were sacrificed during or at termination of the study. At 5,000 ppm, obvious nervous system effects (e.g., incoordination, lethargy) were most apparent in dogs and also observed in monkeys and mice. Rats did not demonstrate any of these sedative effects. At 1,000 ppm, these effects were observed to a lesser extent in monkeys and mice, but dogs still displayed prominent CNS depressive behavior. Histopathologic analysis revealed edema of the brain in three dogs that died during exposure to 5,000 ppm dichloromethane. No other gross brain-related changes were reported. The results indicate that continuous exposure to 1,000 ppm was an adverse effect level for mortality and effects on the nervous system and liver in dogs (exposed for up to 4 weeks) and for BW changes in rats (exposed for 14 weeks). The 5,000 ppm level induced mortality in beagle dogs, ICR mice, and rhesus monkeys (but not Sprague-Dawley rats); obvious nervous system effects in dogs, mice, monkeys, and rats; and gross liver changes in dogs, mice, monkeys, and rats.

In the study by Thomas et al. (1972), female ICR mice were exposed continuously to 0, 25, or 100 ppm dichloromethane for 14 weeks. Spontaneous activity of mice was evaluated by using closed circuit television for monitoring. Mice were evaluated in daily 2-hour testing sessions. The 25 and 100 ppm exposure groups were tested for 2 weeks prior to the onset of dichloromethane exposure. Starting at week 9, mice exposed to 25 ppm dichloromethane exhibited increases in spontaneous activity, but no quantitative measurements or statistical analysis were reported. The authors stated that no significant effect was observed in the group exposed to 100 ppm.

FOB—subchronic (13 week) study

Only one study, a 13-week inhalation study in F344 rats (Mattsson et al., 1990) has conducted an FOB testing paradigm following a subchronic exposure to dichloromethane. Groups of rats (12/sex/exposure level) were exposed to 0, 50, 200, or 2,000 ppm dichloromethane 6 hours/day, 5 days/week for 13 weeks. An additional group of rats was exposed to 135 ppm CO to induce approximately 10% COHb, approximately the level produced by saturation of oxidative metabolism of dichloromethane. After the 13 weeks of exposure (beginning 65 hours after the last exposure), rats were subject to an FOB to evaluate any

neurobehavioral changes from the dichloromethane exposure. Autonomic parameters were first characterized and then the rat was placed in a clear plastic box to evaluate locomotor activity and then responsiveness to touch, sharp noise, and tail pinch. Hind-limb grip strength was also measured by using a strain gauge. All animals were examined clinically at weekly intervals and were tested at the end of the exposure period by FOB, grip strength, BW, temperature, and sensory-evoked potentials. No exposure-related effects were observed on the FOB, grip strength, or sensory-evoked potentials. No histopathologic changes were noted in brains, spinal cords, or peripheral nerves from the high-dose dichloromethane group compared with control animals. In the absence of changes, lower concentrations were not examined.

Learning and memory—acute study

In a study by Alexeef and Kilgore (1983), a learning and memory evaluation was conducted following acute exposure to dichloromethane. Mice were exposed to 168 mg/L (~47,000 ppm) dichloromethane and were tested for learning ability by using a passive-avoidance conditioning task. Male Swiss-Webster mice (3, 5, and 8 weeks old) were used in this study. In the passive avoidance task, mice were placed on a metal platform that extended into a hole. If the mouse went into the hole (a darkened area, which would be the preferred area for the mouse), it received a foot shock. Prior to the training session, mice were exposed to either air or ~47,000 ppm dichloromethane. Animals were exposed to dichloromethane until there was a loss of the righting reflex, which would take about 20 seconds on average, and then placed back in their home cage. One hour after exposure, animals were trained to learn the passive avoidance task. A mouse was considered to have learned the task once it remained on the platform for at least 30 seconds without entering the hole. Mice were then tested for recollection of the task at either 1, 2, or 4 days after the initial training session. In the learning phase of the task, 74% of the control mice retained the task in comparison to 59% of the dichloromethane-exposed group, indicating the significant effect of dichloromethane on learning. There was also an age-related effect since exposed 3-week-old mice were less likely to recall the task than five- or eight-week-old mice. There was no difference in task recall between the 5- and 8-week-old mice. Dichloromethane, at the exposure used in the study, was demonstrated to be non-analgesic, since pain-response times were comparable to those in air-exposed animals in the hot-plate pain test, and therefore the results of the passive avoidance test were not confounded by potential analgesic effects. As a result, it is demonstrated that exposure to an acute and high concentration of dichloromethane alters learning ability in mice.

4.4.3.2.2. ***Inhalational exposure—neurophysiological studies.*** The effect of dichloromethane on sensory stimuli was evaluated by measuring sensory-evoked responses during an acute exposure (Rebert et al., 1989) and following a subchronic (13 week) exposure (Mattsson et al., 1990). Rebert et al. (1989) evaluated the effects of dichloromethane on sensory-evoked

potentials (auditory, visual, and somatosensory) in F344 rats exposed to 0, 5,000, 10,000, and 15,000 ppm dichloromethane for 1 hour in a head-only exposure chamber. Twelve adult male rats were implanted with chronic epidural electrodes placed over the visual and somatosensory cortices. Each rat served as its own control, with a 1-week recovery period between testing sessions. During each testing session, spontaneous electroencephalograms were recorded. Additionally, brainstem-auditory-evoked responses (BAERs) (tone stimulus), cortical-auditory-evoked potentials (CAEPs) (click stimulus), FEPs (flash stimulus), and somatosensory-evoked potentials (SEPs) (tail current stimulus) were measured in response to the stimuli. Dichloromethane decreased the SEP response to the tail current stimulus, and earlier components of the FEP response were attenuated and eventually eliminated with increasing exposures. The BAER response profile was also significantly altered. Dichloromethane completely abolished the CAEP at all concentrations tested. Slight recovery of this response was noted approximately 1 hour after exposure. The collective results strongly suggest a CNS depressive profile for dichloromethane and indicate that this chemical affects the auditory, visual, and somatosensory regions of the brain.

In a subchronic exposure study, male and female F344 rats were exposed to dichloromethane for 6 hours/day, 5 days/week for 13 weeks (Mattsson et al., 1990). Twelve animals of each sex were selected for exposure to 0, 50, 200, or 2,000 ppm dichloromethane or 135 ppm CO. For electrophysiological measures, rats were surgically implanted with epidural electrodes 10 weeks after the onset of exposure. Electrodes were placed over the somatosensory, visual, and cerebellar region. Electrophysiological measures that were recorded included FEP measurements, cortical flick fusion responses, CAEPs, BAERs, and SEPS recorded from the sensory (SEP-S) and cerebellar (SEP-C) regions. None of these measures were significantly altered by any dichloromethane or CO treatment in this study. However, it should be noted that all of the electrophysiological measures were conducted at least 65 hours after the last dichloromethane exposure. As a result, it can be concluded that a subchronic exposure to dichloromethane did not result in persistent changes in any of the neurophysiological measures that were evaluated in this study. It is not known if any neurological compensation occurred since SEP measurements were not taken during actual dichloromethane exposure in this subchronic study.

Based on these two studies, the significant changes noted in several SEP measures during dichloromethane exposure were not observed after a subchronic exposure where animals were tested at least 65 hours after the last exposure. As a result, it is difficult to ascertain if tolerance is developed to the dichloromethane-mediated changes in sensory potentials during an acute exposure or if these effects are still maintained during repeated exposure, since measurements were not taken during the subchronic exposure.

4.4.3.2.3. ***Inhalational exposure—neurochemistry and neuropathology studies.*** The studies evaluating specific neurochemical changes in relation to dichloromethane exposure include studies of effects of short-term (3-day to 2-week) exposures (Fuxe et al., 1984; Savolainen et al., 1981) and subchronic (3-month) exposures (Karlsson et al., 1987; Briving et al., 1986; Rosengren et al., 1986).

Savolainen et al. (1981) examined three different exposure schemes in male Wistar rats. The rats were exposed to 500, 1,000, or 1,000 ppm TWA dichloromethane for 6 hours/day, 5 days/week for 2 weeks. (Note: The abstract of this paper describes the exposures as 500, 1,000, and 100 ppm TWA, but, based on information in the body of the paper, the abstract appears to be incorrect.) The 1,000 ppm TWA exposure consisted of a basal 100 ppm exposure with two 2,800 ppm 1-hour peak concentrations (at 1 and 4 hours) resulting in a time-weighted exposure of 1,000 ppm. Brains were removed from rats at the end of study and analyzed. The 1,000 ppm TWA group displayed increases in cerebral RNA. Other changes noted for this group in the cerebrum included significant increases in NADPH diaphorase and succinate dehydrogenase activity. In the 1,000 ppm constant exposure group, acid proteinase activity was below the levels observed in control animals in the first week but increased to levels above control animals in the second week. In the cerebellum, there were no changes in RNA concentration, and there was a decrease in succinate dehydrogenase activity in both the 1,000 and 1,000 ppm TWA groups. After a 7-day withdrawal, RNA levels in the cerebrum were significantly lower in the 1,000 ppm group. Succinate dehydrogenase levels remained lowered in the 1,000 ppm TWA group after the 7-day exposure-free period. No significant effects were seen at 500 ppm.

Fuxe et al. (1984) evaluated changes in brain catecholamine levels after a 3-day exposure to dichloromethane, using male Sprague-Dawley rats. Rats were exposed to 70, 300, and 1,000 ppm dichloromethane 6 hours/day for 3 consecutive days. Additional groups of rats were exposed to the same levels of dichloromethane and given intraperitoneal injections of the tyrosine hydroxylase inhibitor, α-methyl-dl-*p*-tyrosine methyl ester (H44/68), 2 hours prior to sacrifice. Brains were removed, stained, and evaluated for catecholamine changes 16–18 hours after the last exposure. Catecholamine levels were measured in the hypothalamus, frontal cortex, and caudate nucleus among other brain regions. At all exposures, there was a significant decrease by approximately 10–15% of catecholamine concentrations in the posterior periventricular region of the hypothalamus. In the medial part of the caudate nucleus, which is involved in memory processes, catecholamine levels were significantly higher (12%) in the 70 ppm group but significantly lower in the 300 ppm (1%) and 1,000 ppm (8%) groups compared with controls. The impact of dichloromethane was also evaluated on the hypothalamic-pituitary gonadal axis. The hypothalamus regulates secretion of reproductive hormones, such as follicle-stimulating hormone and luteinizing hormone. The levels of the hormone release were not significantly changed with dichloromethane exposure. However, when rats were dosed concurrently with H44/68 and dichloromethane, statistically significant

inversely dose-related increases in luteinizing hormone levels were observed (330, 233, and 172% higher than controls in the 70, 300, and 1,000 ppm groups, respectively). The study overall demonstrates significant changes in catecholamine levels in the hypothalamus and caudate nucleus. No significant changes in catecholamine levels in the frontal cortex were reported. Catecholamine level changes in the hypothalamus did not appear to significantly affect hormone release; however, decreased catecholamine levels in the caudate nucleus at higher exposures may lead to memory and learning impairment.

A series of studies were conducted in male and female Mongolian gerbils exposed continuously to 210 ppm (Karlsson et al., 1987; Briving et al., 1986), 350 ppm, or 700 ppm (Rosengren et al., 1986) dichloromethane for 3 months, followed by a 4-month exposure-free period. High mortality rates occurred at 350 ppm (6/10 males and 3/10 females by 71 days) and 700 ppm (10/10 males and 9/10 females by 52 days). Rosengren et al. (1986) monitored two astroglial proteins, S-100 and GFA, as well as DNA concentrations in the brain. Decreased DNA concentrations were noted in the hippocampus at both the 210 and 350 ppm exposures. At 350 ppm, there was also decreased DNA concentration in the cerebellar hemispheres, indicating a decreased cell density in these regions, probably due to cell loss. Increased astroglial proteins were found in the frontal and sensory motor cerebral cortex, which directly correlated to the astrogliosis that was observed in those areas. Up-regulation of these astroglial proteins is a good indicator of neuronal injury (Rosengren et al., 1986).

Karlsson et al. (1987) measured DNA concentrations in different regions of the gerbil brain. After the solvent-free exposure period, brains were removed and the olfactory bulbs and cerebral cortices were dissected. Brain weights and weights of the dissected brain regions were the same between control and dichloromethane-exposed animals. The total protein concentration per wet weight was not significantly different between dichloromethane-exposed and control animals. However, DNA concentrations per wet weight were significantly decreased in the hippocampus after dichloromethane exposure. No other examined regions demonstrated significant changes in DNA concentrations after dichloromethane exposure. This selective DNA concentration decrease observed in the hippocampus is a sign of neurotoxicity and may possibly explain why some studies have noted memory and learning deficits with dichloromethane exposure. In a companion paper, in which only the 210 ppm level was tested, it was found that exposure to dichloromethane decreased the levels of glutamate, γ-aminobutyric acid, and phosphoethanolamine in the frontal cortex, while glutamine and γ-aminobutyric acid were increased in the posterior cerebellar vermis (Briving et al., 1986). Increased levels of glutamate in the posterior cerebellar vermis could reflect an activation of astrocytic glia, since glutamine synthetase is localized exclusively in astrocytes. The gerbils did not have a solvent-free exposure period as in the other two studies (Karlsson et al., 1987; Rosengren et al., 1986). The exposure regime in these studies did not affect BW or brain weight. Furthermore, the neurochemical changes observed in these studies were not attributed to formation of CO.

Neurological changes have been investigated by measuring changes in neurotransmitter levels and changes in neurotransmitter localization. Changes in catecholamine levels in the caudate nucleus after an acute exposure (Fuxe et al., 1984) as well as decreased DNA content in the hippocampus after a subchronic dichloromethane exposure (Rosengren et al., 1986) suggest that memory functions are altered since both brain regions are associated with learning and memory. The results from Fuxe et al. (1984) directly correlated with the finding that learning and memory were impaired in mice after an acute (single) and very high exposure (47,000 ppm) to dichloromethane (Alexeef and Kilgore, 1983). Additionally, changes in the hippocampus also suggest memory effects after a long-term, continual exposure to dichloromethane, although no conclusive evidence has been presented to date. In another subchronic, continuous exposure to 350 ppm dichloromethane for 3 months, decreased DNA concentration was observed in the cerebellar hemispheres of Mongolian gerbils and is suggestive of cell loss (Rosengren et al., 1986). However, in a 2-week exposure study in male Wistar rats, RNA changes were not noted in the cerebellum, although enzyme activity was significantly decreased in this region (but was increased in the cerebrum) (Savolainen et al., 1981). These results suggest that the cerebellum is a target for dichloromethane. Noted neurobehavioral effects that may be linked to impaired cerebellar function include changes in motor activity and impaired neuromuscular function (Moser et al., 1995).

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

4.5.1. Genotoxicity Studies

4.5.1.1. *In Vitro Genotoxicity Assays*

Bacterial, Yeast, and Fungi mutagenicity assays

Numerous in vitro studies have demonstrated dichloromethane as being mutagenic in bacterial assays, yeast, and fungi, and several studies provide evidence that the genotoxic action of dichloromethane in bacterial systems is enhanced in the presence of GSH (e.g., Dillon et al., 1992; Their et al., 1993; Oda et al., 1996; DeMarini et al., 1997; Pegram et al., 1997) (Table 4-29). Considering the results are primarily dependent on the presence of GSH, activation likely involves the GST-T1 metabolic pathway, which produces two proposed DNA-reactive metabolites, S-(chloromethyl)glutathione and formaldehyde.

Dichloromethane induced mutations in *Salmonella typhimurium* strains containing GSH (e.g. TA100, TA98). These effects were not markedly influenced by the addition of exogenous mammalian liver fractions, suggesting that endogenous metabolism in these strains was sufficient to activate dichloromethane (Green, 1983; Jongen et al., 1982; 1978; Gocke et al., 1981). In support of this hypothesis, dichloromethane exposure of NG-11, a glutathione-deficient variant of *S. typhimurium* strain TA100, produced twofold fewer base-pair mutations

Table 4-29. Results from in vitro genotoxicity assays of dichloromethane with bacteria, yeast, or fungi

Assay	Test system	Concentration(s)	Results: Without metabolic activation –S9	Results: With metabolic activation +S9	Reference
		Bacteria			
Reverse mutation	*S. typhimurium* TA98[a], TA100[a]	6-hour exposure to 0, 7,000, and 14,000 ppm	+	+	Jongen et al. (1978)
Reverse mutation	*S. typhimurium* TA98, TA100	Up to 3,600 µg/plate	+	++	Gocke et al. (1981)
Reverse mutation	*S. typhimurium* TA1535[b], TA1537[b], TA1538[b]	Up to 3,600 µg/plate	–	–	Gocke et al. (1981)
Reverse mutation	*S. typhimurium* TA100	6-hour exposure to 0, 7,000, and 14,000 ppm	+	++	Jongen et al. (1982)
Reverse mutation	*S. typhimurium* TA100	Up to 84,000 ppm, 3-day exposure	+	+	Green (1983)
Reverse mutation	*S. typhimurium* TA100, TA1535, TA1950[a], *E. coli* WU361089[a]	10 µL/plate	+ for TA100, TA1950, WU361089 – for TA1535	Not determined	Osterman-Golkar et al. (1983)
Reverse mutation	*S. typhimurium* TA100		+	Not determined	Zeiger (1990)
Reverse mutation	*S. typhimurium* TA100, NG54[c]	2- and 6-hour exposures to 0, 2,500, 5,000, 7,500, and 10,000 ppm	+	+	Dillon et al. (1992)
Reverse mutation	*S. typhimurium* TA100, TA1535 and TA1538 (+GSTA 1-1 and GSTP 1-1)	0, 50, 100, and 200 µL/plate	+ for TA100 – for TA1535, TA1538	Not determined	Simula et al. (1993)
Reverse mutation	*S. typhimurium* TA1535 (+GST 5–5), TA1535 (wild type)	0–2.0 mM/plate	+ for TA1535 (+GST 5–5) – for TA1535 (wild type)	Not determined	Pegram et al. (1997); Thier et al. (1993)
Reverse mutation	*S. typhimurium* TA100, TA100/NG-11[d]	0, 30, 60, 130 mM/plate	++ for TA100 + for TA100/NG-11	Not determined	Graves et al. (1994a)
Reverse mutation	*S. typhimurium* TA100, RSJ100[c]	Up to 24,000 ppm	+ for TA100 + for RSJ100	+ for TA100 + for RSJ100	DeMarini et al. (1997)

Table 4-29. Results from in vitro genotoxicity assays of dichloromethane with bacteria, yeast, or fungi

Assay	Test system	Concentration(s)	Results		Reference
			Without metabolic activation –S9	With metabolic activation +S9	
Forward mutation	*S. typhimurium* BA13	0–130 µmol/plate	+++	+	Roldán-Arjona and Pueyo (1993)
Gene mutation	*S. typhimurium* TA1535/pSK1002[c], NM5004[c]	0, 2.5, 5.0, 10, 20 mM	+ NM5004 – TA1535/ pSK100 2	Not determined	Oda et al. (1996)
Prophage induction	*E. coli* K-39 (λ)	10 µL/plate	+	Not determined	Osterman-Golkar et al. (1983)
Reverse mutation	*E. coli* WP2 uvra pKM101	2- and 6-hour exposures to 6,300, 12,500, 25,000, and 50,000 ppm	+	+	Dillon et al. (1992)
Forward mutation	*E. coli* K12	0, 30, 60, 130 mM/plate	–	+	Graves et al. (1994a)
Forward mutation	*E. coli* Uvr^+, $UvrB^-$	20,000 ppm	+	Not determined	Zielenska et al. (1993)
		Fungi and yeasts			
Mitotic segregation	*Aspergillus nidulans*	Up to 8,000 ppm	+ only at 4,000 ppm; no dose-response relationship established	Not determined	Crebelli et al. (1988)
Gene conversion and recombination	*Saccharomyces cerevisiae*	Up to 209 mM	+	Not determined	Callen et al. (1980)

[a] bacterial strains that have GSH (e.g. TA100, TA 98)
[b] bacterial strains that do not have GSH (e.g. TA1535)
[c] bacterial strains engineered to have more GSH activity than wild type
[d] bacterial strains engineered to have less GSH activity than wild type

compared with exposure of strain TA100, which produces normal levels of GSH. Furthermore, this difference was not apparent when the culture medium contained 1 mM GSH (Graves et al., 1994a).

In contrast to strain TA100, *S. typhimurium* strains TA1535, TA1537, and TA1538 (strains deficient in GSH) did not develop base-pair mutations in response to dichloromethane exposure (Gocke et al., 1981; Osterman-Golkar et al., 1983; Simula et al., 1993; Thier et al., 1993; Pegram et al., 1997). However, when strain TA1535 was transfected with rat GST-T1, dichloromethane induced base-pair reverse mutations (DeMarini et al., 1997; Pegram et al., 1997; Thier et al., 1993). A 60-fold higher concentration of dichloromethane was needed to induce a response (i.e., a sixfold increase over background levels in reverse mutations) in *S. typhimurium* strain TA100 than in TA1535 transfected with rat GST-T1 (DeMarini et al., 1997). This study also included several trihalomethanes; dichloromethane was several fold less genotoxic than dibromochloromethane or bromoform, but was similar in potency to bromodichloromethane (DeMarini et al., 1997; Pegram et al., 1997). The authors suggest that these results support a role of GST-T1 in the mutagenicity of the trihalomethanes.

The mutagenic effects of dichloromethane have also been examined in fungi and yeasts assays with both systems reporting positive results. Fungi assays were positive for mitotic segregation in *Asperigillus ridulans* (Crebelli et al., 1988) but there was not a dose response relationship as only the 4,000 ppm dichloromethane exposure was positive (exposure up to 8,000 ppm). A yeast assay was positive for gene conversion and recombination in *Saccharomyces cerevisiae* for concentrations up to 209 mM (Callen et al., 1980).

Mammalian assays

In the in vitro mammalian system studies conducted with murine cell lines (Table 4-30), dichloromethane was negative for producing point mutations in the mouse lymphoma L5178Y cell line (Thilagar et al., 1984), but was positive in producing single stranded DNA breaks in mouse Clara cells (Graves et al., 1995) and mouse hepatocytes (1994b). Given that exposure to dichloromethane results specifically in lung and liver tumors, this pattern is not surprising. Additionally, GST is localized in the nucleus of hepatocytes and lung cells in the mouse (Mainwaring et al., 1996), which would also increase sensitivity of these particular cell fractions to genotoxic effects of dichloromethane. DNA single strand breaks (SSBs) were induced at lower concentrations in mouse hepatocytes (0.5 mM) than in rat hepatocytes (30 mM). The extent of DNA damage was shown to be reduced to the background level seen in control (no exposure) conditions by pretreating the cells with buthionine sulfoxime to deplete cellular levels of GSH and thus inhibit dichloromethane metabolism via the GST pathway (Graves et al. 1995; 1994b). Similar results were seen in mouse lung Clara cells. Freshly isolated Clara cells from the lungs of $B6C3F_1$ mice also showed significantly increased, concentration-dependent amounts of DNA SSBs when incubated in vitro for 2 hours in the presence of 5–60 mM dichloromethane.

Pretreatment with buthionine sulphoximine before Clara-cell isolation or the presence of buthionine sulphoximine in the culture medium decreased the amount of in vitro DNA damage induced.

In a series of experiments with freshly isolated hepatocytes from multiple species (Table 4-30), DNA-protein cross-links were detected in hepatocytes of $B6C3F_1$ mice but not in hepatocytes of F344 rats, Syrian golden hamsters, or three human subjects, following 2-hour in vitro exposure to concentrations ranging from 0.5–5 mM dichloromethane (Casanova et al., 1997). Within the range of concentrations tested, DNA-protein cross-links in mouse hepatocytes appeared to increase with increasing concentration of dichloromethane.

Negative results for dichloromethane were predominantly seen in in vitro test systems that used rat or hamster cell lines with low or no GST activity (Table 4-30). Several genotoxic endpoints, including DNA and protein synthesis (Garrett and Lewtas, 1983), chromosomal aberrations or sister chromatid exchanges (Thilagar et al., 1984; Thilagar and Kumaroo, 1983; Jongen et al., 1981), unscheduled DNA synthesis (Thilagar et al., 1984; Andrae and Wolff, 1983; Jongen et al., 1981), and mutations (Thilagar et al., 1984; Jongen et al., 1981) were evaluated in these cell lines. In contrast, positive results (DNA-protein cross links and DNA SSBs) were observed when mouse liver cytosol was included in Chinese hamster ovary (CHO) cells (Graves et al., 1994b; Graves et al., 1995). Dichloromethane also induced hypoxanthine-guanine phosphoribosyl transferase (HPRT) gene mutations in CHO cells when they were incubated with GST-competent mouse liver cytosol preparations (Graves et al., 1996).

The instability of the S-(chloromethyl)glutathione-adducts presents considerable challenges to studies of these products (Hashmi et al., 1994). Kayser and Vuilleumier (2001), however, demonstrated the formation of DNA adducts with radiolabeled dichloromethane in calf thymus DNA in the presence of dichloromethane dehalogenase/GST purified from a bacterial source (*Methylophilus* sp. strain DM11) and GSH (Table 4-30). The type of adduct could not be identified because of low yield, but it was determined that guanine was more actively incorporated than cytosine, adenine or thymine by at least 2 fold in the presence of GST-activated dichloromethane, indicating a base specificity for these adducts. Incubation of calf thymus DNA with formaldehyde and GSH, however, did not result in detectable DNA adduct formation. In another study, Marsch et al. (2004) further evaluated the presence of adducts in calf thymus DNA in the presence of dichloromethane and human (GSTT1-1), rat (GST 5-5) or bacterial (DM11) GST (Marsch et al., 2004). This study found that all three enzymes yielded a similar pattern of adduct formation, forming primarily with guanine and to a lesser extent with cytosine, adenine, and thymine (2-3 fold less than guanine), consistent with the results reported by Kayser and Vuilleumier (2001). High levels of guanosine-specific adducts were also seen with S-(1-acetoxymethyl)glutathione, a compound that is structurally similar, but more stable, than S-(chloromethyl)glutathione (Marsch et al., 2001). These findings indicate that the S-(chloromethyl)glutathione intermediate formed by GSH conjugation has mutagenic potential and

is likely responsible, at least in part, for the mutagenic response observed following dichloromethane exposure.

In studies with human cell lines or isolated cells, positive results were reported for sister chromatid exchanges and chromosomal aberrations (Thilagar et al., 1994) and in the micronucleus test (Doherty et al., 1996). Negative results with human cells were seen in the unscheduled DNA synthesis assays (Perocco and Prodi, 1981; Jongen et al., 1981), DNA SSBs, and DNA-protein cross-links (Graves et al., 1995; Casanova et al., 1997).

Dichloromethane-induced DNA damage (comet assay) was examined in primary cultures of human lung epithelial cells collected by brush biopsy from four healthy volunteers (Landi et al., 2003). This study was designed to assess the genotoxicity of four thrihalomethanes (chloroform, bromodichloromethane, dibromochloromethane and bromoform), with dichloromethane included because of its known activation by GST-T1. Two of the subjects were of the GST-T1^{+} genotype, and two were of the GST-T1^{-} genotype.[6] The cells had been frozen, and GST activity was not detected in the cultured cells. DNA damage was reported to occur in the combined GST-T1^{-} samples (tail extent moment 7.1, 13.7 and 15.3 in the 10, 100 and 1000 μM dichloromethane groups, respectively), but not in the combined GST-T1^{+} samples (tail extent moment 8.1, 11.5 and 10.4 in the 10, 100 and 1000 μM dichloromethane groups, respectively). This pattern was not seen across the individual samples, however, as only one sample exhibited a clear dose-response gradient. Given the absence of GST activity, an analysis combining the four samples could provide a more informative picture of the dose-response relation between dichloromethane (and the other compounds) studied and DNA damage. For dichloromethane, values of 9.4, 7.6, 12.6, and 12.9 were seen in the 0, the 10, 100 and 1000 μM groups, respectively. This pattern was similar to that seen with chloroform (9.4, 6.9, 11.4, and 12.7 in the 0, the 10, 100 and 1000 μM groups, respectively), but weaker than the pattern for bromoform (9.4, 12.5, 15.8, and 18.2 in the 0, the 10, 100 and 1000 μM groups, respectively), and much weaker than for bromodichloromethane (9.4, 25.2, 28.5, and 39.1 in the 0, the 10, 100 and 1000 μM groups, respectively).[7] No dose-response gradient was seen with dibromochloromethane (9.4, 6.5, 8.1 and 8.0 in the 0, the 10, 100 and 1000 μM groups, respectively). This relative pattern is also seen in the estimated slopes (beta coefficient for the change in tail extent moment per unit increase in μM concentration): 0.0, 0.003, 0.004., 0.006 and 0.02 for dibromochloromethane, dichloromethane, chloroform, bromoform, and bromodichloromethane, respectively (statistical significance not reported).

[6] Landi et al. (2003) did not clearly describe their treatment of GST-T1$^{+/-}$ heterozygote genotypes; the EPA believes it is most likely they were included in the pool from which the GST-T1^{+} samples were drawn. In addition, there is a discrepancy in the paper regarding this the coding of the GST-T1 genotypes. Samples A and C are noted to be the GST-T1^{-} samples in one part of the paper, and C and D are described as the GST-T1^{-} samples in another part of the paper.

[7] These values are based on the mean of the GST-T1^{+} and the GST-T1^{-} samples from Table 1 of Landi et al. (2003)

A stronger and more consistent response was seen under the same experimental conditions with bromodichloromethane, but dibromochloromethane resulted in no increase in DNA damage in any of the donor cells at any concentration tested.

Several studies have examined patterns of mutations or DNA damage with dichloromethane and formaldehyde to assess the relative role of S-(chloromethyl)glutathione and formaldehyde in the observed genotoxicity. In a study in CHO cells incubated with dichoromethane (0.3% plus mouse liver cytosol), 2.5-fold increases in DNA-protein cross-links that are indicative of formaldehyde exposure were observed, compared with a 25-fold increase when 1 mM formaldehyde was added directly to cultures. Both treatments induced a comparable degree of DNA SSBs (Graves and Green, 1996). In a subsequent study, Graves et al. (1996) compared the mutational spectra induced by dichloromethane to that induced by direct addition of formaldehyde or 1,2-dibromoethane (a chemical known to act through a glutathionyl conjugate metabolite) at the HPRT locus in CHO cells. The mutations induced by dichloromethane and 1,2-dibromoethane were predominantly GC to AT transitions, while all six formaldehyde-induced mutants sequenced were single base transversions. This provided further evidence that the S-(chloromethyl)glutathione intermediate may be primarily responsible for dichloromethane genotoxicity. In contrast, Hu et al. (2006) found evidence of significant amounts of formaldehyde formation following dichloromethane exposure in the cytosol of V79 (hamster) cells transfected with the murine GSTT1 gene compared to the parent cell line. In accordance with this, they observed concentration-dependent increases in DNA-protein crosslinks in the GSTT1 transfected cells using the comet assay with and without proteinase K treatment that frees DNA from crosslinks and allows DNA migration. These findings are consistent with those by Casanova et al. (1997), who performed a comparison of the amounts of DNA-protein and RNA-formaldehyde crosslinks formed following dichloromethane exposure in hepatocytes isolated from mice, rats, hamsters, and human GSTT1 genetic variants. Only DNA-protein crosslinks were observed in mouse hepatocytes, but RNA-formaldehyde crosslinks were found in all species, which were highest in the mouse hepatocytes, and were 4-, 7-, and 14-fold higher than rats, humans, and hamsters. These results showed that human hepatocytes can metabolize dichloromethane to formaldehyde, resulting in RNA-formaldehyde crosslinks. In addition, the results indicate, that there is considerable variation among species, and that the human variation in the GSTT1 gene can also affect the amount of formaldehyde produced. The authors also noted that comparing results following ectopic addition of formaldehyde directly to cells with results following dichloromethane metabolism in situ can be misleading, as the formaldehyde produced internally may reside in different locations intracellularly, potentially affecting the capability of interacting with DNA. These results show that, while most studies indicate the importance of the S-(chloromethyl)glutathione intermediate in mediating genotoxic damage following dichloromethane exposure, DNA damage resulting from formaldehyde formation should also be considered.

Table 4-30. Results from in vitro genotoxicity assays of dichloromethane with mammalian systems, by type of test

Assay	Test system	Concentrations	Results	Reference
		Mouse		
Point mutation	Mouse lymphoma L5178Y cells	Not provided	Negative	Thilagar et al. (1984)
DNA SSBs by alkaline elution	Mouse hepatocytes	0, 0.4, 3.0, 5.5 mM	Positive at 0.4 mM	Graves et al. (1994b)
DNA SSBs by alkaline elution	Mouse Clara cells	0, 5, 10, 30, 60 mM	Positive, but DNA damage was reduced by incubating in the presence of GSH depletory	Graves et al. (1995)
DNA-protein cross-links	Mouse hepatocytes	0.5–5 mM	Positive	Casanova et al. (1997)
		Rat		
Unscheduled DNA synthesis	Rat hepatocytes	Up to 16 mM (measured); 30 mM (nominal)	Negative	Andrae and Wolff (1983)
Unscheduled DNA synthesis	Rat hepatocytes	Not provided	Marginally positive	Thilagar et al. (1984)
DNA SSBs by alkaline elution	Rat hepatocytes	0, 30, 90, 90 mM	Positive at 30 mM	Graves et al. (1994b)
DNA-protein cross-links	Rat hepatocytes	0.5–5 mM	Negative	Casanova et al. (1997)
		Hamster with GST activity from mouse		
DNA-protein cross-links	Chinese hamster ovary cells	60 mM	Positive with mouse liver cytosol (negative without) at much higher concentrations of dichloromethane (60 mM) than formaldehyde (0.5–4 mM).	Graves et al. (1994b)
HPRT[a] mutation analysis	Chinese hamster ovary cells	2,500 ppm	Positive with mouse liver cytosol	Graves et al. (1996)
DNA SSBs and DNA-protein cross-links	Chinese hamster ovary cells	3,000 ppm (0.3%, volume per volume [v/v]) and 5,000 ppm (0.5%, v/v)	Positive at concentration of 0.5% (v/v) for SSBs in presence of mouse liver cytosol, but increase in DNA-protein cross-links marginal; formaldehyde (in absence of mouse liver cytosol) was positive at 0.5 mM for both DNA SSBs and DNA-protein cross-links; Chinese hamster ovary cell cultures were suspended.	Graves and Green (1996)

Table 4-30. Results from in vitro genotoxicity assays of dichloromethane with mammalian systems, by type of test

Assay	Test system	Concentrations	Results	Reference
DNA-protein cross-links	Syrian golden hamster hepatocytes	0.5–5 mM	Negative	Casanova et al. (1997)
Comet Assay	V79 hamster cells transfected with mouse GSTT1	2.5, 5, 10 mM	A significant, dose-dependent increase in DNA damage resulting from DNA-protein crosslinks in V79 cells transfected with mouse GSTT1 compared to parental cells	Hu et al. (2006)
		Hamster without GST activity from mouse		
Forward mutation	Chinese hamster epithelial cells	5,000, 10,000, 30,000, 50,000 ppm	Negative	Jongen et al. (1981)
Unscheduled DNA synthesis	Chinese hamster epithelial cells	5,000, 10,000, 30,000, 50,000 ppm	Negative	Jongen et al. (1981)
Sister chromatid exchange	Chinese hamster epithelial cells	5,000, 10,000, 20,000, 30,000, and 40,000 ppm	Weak positive with or without rat-liver microsomal system	Jongen et al. (1981)
Chromosomal aberrations	Chinese hamster ovary cells	Not provided	Positive, independent of rat liver S9	Thilagar and Kumaroo (1983)
Sister chromatid exchange	Chinese hamster ovary cells	Not provided	Negative with or without rat liver S9	Thilagar and Kumaroo (1983)
DNA and protein synthesis	Chinese hamster ovary cells	Up to 1,000 μg/mL	Negative	Garrett and Lewtas (1983)
DNA SSBs by alkaline elution	Hamster hepatocytes	0.4–90 mM	Negative	Graves et al. (1995)
		Calf		
DNA Adducts	Calf thymus DNA	50 mM	Positive in the presence of bacterial GST DM11 and dichloromethane dehalogenase. Adducts primarily formed with the guanine residues.	Kayser and Vuilleumier (2001)
DNA Adducts	Calf thymus DNA	0 -8.0 umol (0 – 60 mM)	Positive in the presence of bacterial GST DM11, rat GST5-5, and human GSTT1-1. Adducts primarily formed with the guanine residues.	Marsch et al. (2004)
		Human		
Unscheduled DNA synthesis	Human peripheral lymphocytes	250, 500, 1,000 ppm	Negative with or without rat liver S9	Perocco and Prodi (1981)
Unscheduled DNA synthesis	Primary human fibroblast	5,000, 10,000, 30,000, 50,000 ppm	Negative	Jongen et al. (1981)

Table 4-30. Results from in vitro genotoxicity assays of dichloromethane with mammalian systems, by type of test

Assay	Test system	Concentrations	Results	Reference
Sister chromatid exchange	Human peripheral lymphocytes	Not provided	Weak positive	Thilagar et al. (1984)
Chromosomal aberrations	Human peripheral lymphocytes	Not provided	Positive	Thilagar et al. (1984)
DNA SSBs by alkaline elution	Human hepatocytes	Up to 120 mM	Negative at concentrations between 5 and 120 mM	Graves et al. (1995)
Micronucleus test	Human AHH-1, MCL-5, h2E1 cell lines	Up to 10 mM	Positive in all three cell lines	Doherty et al. (1996)
DNA-protein cross-links	Mouse, rat, hamster, human hepatocytes	0.5–5 mM	Negative	Casanova et al. (1997)
DNA damage by comet assay	Primary human lung epithelial cells	10, 100, 1,000 μM	Weak trend, independent of GST activity (GST enzymatic activity not present in the cultured cells)	Landi et al. (2003)

[a]HPRT = hypoxanthine-guanine phosphoribosyl transferase

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4.5.1.2. *In Vivo Genotoxicity Assays*

Genotoxicity findings in *Drosophila melanogaster* assays are mixed (Table 4-31). A study of gene mutation in *D. melanogaster* showed a marginal increase in sex-linked recessive deaths following oral exposure (Gocke et al., 1981). An additional feeding study (Rodriguez-Arnaiz, 1998) reported a positive response in the somatic w/w^+ assay. A third study of *D. melanogaster* (Kramers et al., 1991) found no evidence of increased sex-linked recessive deaths, somatic mutations, or recombinations following exposure to airborne dichloromethane.

Table 4-31. Results from in vivo genotoxicity assays of dichloromethane in insects

Assay	Test system	Doses	Result	Reference
Gene mutation (sex-linked recessive lethal)	Drosophila	125, 620 mM	Positive (feeding exposure)	Gocke et al. (1981)
Gene mutation (sex-linked recessive lethal, somatic mutation and recombination)	Drosophila	6 hours—1,850, 5,500 ppm 1 week—2,360, 4,660 ppm 2 weeks—1,370, 2,360 ppm (all approximate)	Negative (inhalation exposure)	Kramers et al. (1991)
Somatic w/w+ assay	Drosophila	50, 100, 250, 500 mM	Positive (feeding exposure)	Rodriguez-Arnaiz (1998)

Some in vivo studies investigating certain genotoxic endpoints in mice exposed to dichloromethane produced negative results (Table 4-32). Unscheduled DNA synthesis was not induced in hepatocytes from mice (and rats) after 2- or 6-hour inhalation exposure to concentrations that were carcinogenic in the NTP (1986) mouse bioassay (Trueman and Ashby, 1987) or other exposure routes (Lefevre and Ashby, 1989). Although positive results were not observed in the unscheduled DNA synthesis studies, it is generally recognized that this assay is not sensitive for detecting genotoxic chemicals (Eastmond et al., 2009; Madle et al., 1994). Distinct, unequivocal cytogenetic effects (e.g., induction of micronuclei, sister chromatid exchanges, or chromosome aberrations) were not consistently found in bone marrow or erythrocytes in several studies of mice after acute oral exposures (Sheldon et al., 1987) or parenteral exposures (Westbrook-Collins et al., 1990; Gocke et al., 1981). However, tumorigenic effects in mice are generally localized to the liver and lung (due to high GST activity) and therefore it is not surprising that genotoxic effects were, for the most part, not observed in the bone marrow or erythrocytes (cell types with minimal GST activity). Crebelli et al. (1999) stated that genotoxic effects induced by halogenated hydrocarbons (such as dichloromethane) are not very effective in inducing micronucleus formation in mouse bone marrow and a negative bone marrow micronucleus assay should not offset the consistently positive in vitro results (Dearfield and Moore, 2005)

When genotoxic endpoints were examined in the cancer target tissues (liver and lung) in mice exposed to dichloromethane, positive results were consistently reported (Table 4-32).

These findings provide supporting evidence that GST-pathway metabolites may be key actors in the genotoxic effects and carcinogenic mode of action for dichloromethane. Increased sister chromatid exchanges were found in lung cells and peripheral lymphocytes from mice exposed by inhalation for 2 weeks to 8,000 ppm or for 12 weeks to 2,000 ppm (Allen et al., 1990). Under the same exposure conditions, increased chromosomal aberrations in lung and bone cells and micronuclei in peripheral red blood cells also were found (Allen et al., 1990). DNA-protein cross-links were detected in mouse hepatocytes, but not in lung cells, after a 3-day inhalation exposure to 4000 ppm (Casanova et al., 1992) and between 500 and 4000 ppm (Casanova et al.,1996). DNA damage, detected as increased DNA SSBs, was observed in liver and lung tissue of $B6C3F_1$ mice immediately following 3-hour exposures (Graves et al., 1995). The DNA damage was not detectable 2 hours after in vivo exposure, indicating that DNA repair occurs rapidly. Pretreatment of mice with buthionine sulphoximine, a GSH depletor, caused a decrease, to levels seen in controls, in the amount of DNA damage detected immediately after in vivo exposure in liver and lung tissue, indicating GSH involvement in the genotoxic process. DNA damage (detected by the comet assay) was also reported in liver and lung tissues from male CD-1 mice sacrificed 24 hours after administration of a single oral dose of 1,720 mg/kg of dichloromethane (Sasaki et al., 1998). In this study, DNA damage in lung and liver was not detected 3 hours after dose administration, and no DNA damage occurred at either time point in several other tissues in which a carcinogenic response was not seen in chronic animal cancer bioassays (e.g., stomach, kidney, bone marrow).

Formation of DNA adducts was evaluated in male and female $B6C3F_1$ mice as well as in male F344 rats (Watanabe et al., 2007). Animals were administered 5 mg/kg, i.p., of radiolabeled dichloromethane and sacrificed at 1 or 8 hours after administration. The kidneys and livers were removed and the DNA was isolated from these tissues to evaluate formation of DNA adducts. At the administered dose, DNA adducts were not detected.

Other studies in mice have looked for mutations in specific oncogenes (K-*ras* or H-*ras*) (Devereux et al., 1993) or in a tumor suppressor gene (*p53*) (Hegi et al., 1993) in liver or lung tumors from dichloromethane-exposed mice. These studies have not demonstrated exposure-related patterns of mutations in these genes, although it should be noted that the statistical power of this analysis for the lung tumors is limited (discussed further in sections 4.5.2 and 4.5.3).

Table 4-32. Results from in vivo genotoxicity assays of dichloromethane in mice

Assay	Test system	Route and dose	Duration	Results	Reference
Micronucleus test	Mouse bone marrow	425, 850, or 1,700 mg/kg	2 doses	Negative at all doses	Gocke et al. (1981)
Micronucleus test	Mouse bone marrow	Gavage, 1,250, 2,500, and 4,000 mg/kg	single dose	Negative at all doses	Sheldon et al. (1987)
Micronucleus test	Mouse peripheral red blood cells	Inhalation 6 hr/day, 5 d/wk, 0, 4,000, 8,000 ppm	2 wk	Positive at 4,000 and 8,000 ppm	Allen et al. (1990)
Micronucleus test	Mouse peripheral red blood cells	Inhalation, 6 hr/day, 5 d/wk, 0, 2,000 ppm	12 weeks	Positive at 2,000 ppm	Allen et al. (1990)
DNA synthesis	Mouse liver	Gavage, 1,000 mg/kg; inhalation, 4,000 ppm	single dose; 2 hours	Negative in both oral and inhalation studies	Lefevre and Ashby (1989)
Unscheduled DNA synthesis	Mouse hepatocytes	Inhalation, 2,000 and 4,000 ppm.	2 or 6 hours	Negative	Trueman and Ashby (1987)
Sister chromatid exchange	Mouse bone marrow	Intraperitoneal, 100, 1,000, 1,500, 2,000 mg/kg	single dose	Negative	Westbrook-Collins et al. (1990)
Sister chromatid exchange	Mouse bone marrow	Subcutaneous, 0, 2,500, 5,000 mg/kg	single dose	Negative at all doses	Allen et al. (1990)
Sister chromatid exchange	Mouse lung cells and peripheral lymphocytes	Inhalation 6 hr/day, 5 d/wk, 0, 4,000, 8,000 ppm	2 weeks	Positive at 8,000 ppm	Allen et al. (1990)
Sister chromatid exchange	Mouse lung cells	Inhalation 6 hr/day, 5 d/wk, 0, 2,000 ppm	12 weeks	Positive at 2,000 ppm	Allen et al. (1990)
Chromosome aberrations	Mouse bone marrow	Intraperitoneal, 100, 1,000, 1,500, 2,000 mg/kg	single dose	Negative	Westbrook-Collins et al. (1990)
Chromosome aberrations	Mouse bone marrow	Subcutaneous, 0, 2,500, 5,000 mg/kg	single dose	Negative	Allen et al. (1990)

Table 4-32. Results from in vivo genotoxicity assays of dichloromethane in mice

Assay	Test system	Route and dose	Duration	Results	Reference
Chromosome aberrations	Mouse lung and bone marrow cells	Inhalation, 6 hr/day, 5 d/wk, 0, 4,000, 8,000 ppm	2 weeks	Positive at 8,000 ppm	Allen et al. (1990)
DNA-protein cross-links	Mouse liver and lung cells	Inhalation, 6 hr/day, 3 days, 4,000 ppm	3 days	Positive in mouse liver cells at 4,000 ppm; negative in mouse lung cells	Casanova et al. (1992)
DNA-protein cross-links	Mouse liver and lung cells	Inhalation, 6 hr/day, 150, 500, 1,500, 3,000, 4,000 ppm	3 days	Positive in mouse liver cells at 500–4,000 ppm; negative in mouse lung cells	Casanova et al. (1996)
DNA single strand breaks by alkaline elution	Mouse hepatocytes	Inhalation, 2,000 and 4,000 ppm	3 or 6 hours	Positive at 4,000 ppm at 3 and 6 hours	Graves et al. (1994b)
DNA single strand breaks by alkaline elution	Mouse liver and lung homogenate	Liver: inhalation, 2,000, 4,000, 6,000, 8,000 ppm Lung: inhalation, 1,000, 2,000, 4,000, 6,000 ppm	3 hours 3 hours	Liver: positive at 4,000–8,000 ppm Lung: positive at 2,000–4,000 ppm	Graves et al. (1995)
DNA damage by comet assay	Mouse liver and lung cells	Gavage, 1,720 mg/kg; organs harvested at 0 (control), 3, and 24 hours	single dose	Positive only at 24 hours after dosing	Sasaki et al. (1998)
DNA damage by comet assay	Mouse stomach, urinary bladder, kidney, brain, bone marrow	Gavage, 1,720 mg/kg; organs harvested at 0 (control), 3, and 24 hours	single dose	Negative 3 or 24 hr after dosing	Sasaki et al. (1998)
DNA adducts	Mouse liver and kidney cells	Intraperitoneal, 5 mg/kg	Single dose	Negative	Watanabe et al. (2007)
Kras and Hras oncogenes	Mouse liver and lung tumors	0, 2000 ppm	Up to 104 weeks	No difference in mutation profile between control and dichloromethane-induced liver tumors; number of spontaneous lung tumors (n=4) limits comparison at this site	Devereux et al., 1993
p53 tumor suppressor gene	Mouse liver and lung tumors	0, 2000 ppm	Up to 104 weeks	Loss of heterozygocity infrequently seen	Hegi et al., 1993

Results from in vivo studies in other mammals (i.e., rats and hamsters) of hepatocyte sensitivity to dichloromethane induction of DNA SSBs (Table 4-33) are consistent with interspecies differences in the induction of liver tumors in the inhalation cancer bioassays. A gavage study in rats reported the presence of DNA SSBs with a dose of 1,275 mg/kg (Kitchin and Brown, 1989). The other available studies, however, did not find any genotoxicity following dichloromethane exposure. No increase in unscheduled DNA synthesis in rat hepatocytes was seen following inhalation dichloromethane exposure of 2 to 6 hours at 2000 or 4000 ppm (Trueman and Ashby, 1987), exposure by gavage up to 1000 mg/kg (Trueman and Ashby, 1987), or intraperitoneal exposure of 400 mg/kg (Mirsalis et al., 1989). DNA adducts were not detected in the livers and kidneys of male F344 rats dosed with 5 mg/kg dichloromethane, i.p. (Watanabe et al., 2007). DNA SSBs were significantly increased in hepatocytes isolated from B6C3F_1 mice exposed to 4,831 ppm (4,000 ppm nominal) for 6 hours but were not increased in hepatocytes from Sprague-Dawley rats exposed to 4,527 ppm (4,000 ppm nominal) for 6 hours (Graves et al., 1994b). Results from in vivo interspecies comparisons of dichloromethane induction of DNA-protein cross-links in hepatocytes (expected products of the GSH pathway) are also consistent with the hypothesis that the mouse is more sensitive than other mammalian species due to greater activity of the GST pathway. DNA-protein cross-links were formed in the liver of mice, but not hamsters, following in vivo exposure to air concentrations ranging from 500 to 4,000 ppm, 6 hours/day for 3 days (Casanova et al., 1996). The absence of a genotoxic response in the rat and hamster is consistent with considerably lower GST activity and therefore, these mammalian systems would be expected to be less sensitive at detecting genotoxic effects than the studies conducted in mice.

Table 4-34 compares results from studies of mice and rats in which comparable tissue-specific endpoints were examined in in vivo genotoxicity assays. Several of the endpoints that were positive in mice (e.g., sister chromatid exchange, DNA-protein cross-links, comet assay) have not been examined in the rat. Unscheduled DNA synthesis has been demonstrated in mouse, but not in rat hepatocytes. In contrast to the positive results seen in mouse inhalation exposure studies, DNA SSB induction was not seen in rat inhalation studies but was seen in an oral gavage study.

In summary, the available data provide evidence for mutagenicity of dichloromethane. Most of the in vitro bacterial assays with GST activity showed positive results when there was GST activity. Non-positive results were reported only in bacterial assays with low GST activity; in experiments where GST was added, positive results were then observed. Evaluation of the in vitro mammalian studies also demonstrates consistency of the requirement for GST for observation of genotoxic effects. In rat and hamster cell lines where GST activity is significantly less than mouse, primarily negative results were reported following dichloromethane exposure. However, when mouse liver cytosol or transfected mouse GST were included in these same cell lines, mutagenic effects were reported after dichloromethane exposure. In mouse cell lines,

positive results were obtained in Clara cells but no effects were observed in a mouse lymphoma cell line, which is consistent with the absence of tumors in this site for mice. The results of in vivo mutagenicity in mice also provide support for the site-specificity of the observed tumors. Assays using mouse bone marrow were all negative. However, micronuclei and sister chromatid exchange tests in peripheral blood produced a positive response at high doses. With the exception of one study of unscheduled DNA synthesis in hepatocytes, numerous site-specific studies in either the liver or lung were also positive at various doses. These liver and lung studies included chromosomal aberrations, SSBs and sister chromatid exchanges, and DNA-protein cross-links and correspond to genotoxic and mutagenic effects associated with metabolites from the GST pathway.

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Table 4-33. Results from in vivo genotoxicity assays of dichloromethane in rats and hamsters

Assay	Test system	Route and dose	Duration	Results	Reference
Unscheduled DNA synthesis	Rat hepatocytes	Gavage, 100, 500, 1,000 mg/kg	Liver harvested 4 and 12 hours after dosing	Negative 4 or 12 hours after dosing	Trueman and Ashby (1987)
Unscheduled DNA synthesis	Rat hepatocytes	Inhalation, 2 or 6 hours, 2,000 and 4,000 ppm	2 or 6 hours	Negative at both concentrations and exposure durations	Trueman and Ashby (1987)
Unscheduled DNA synthesis	Rat hepatocytes	Intraperitoneal, single dose, 400 mg/kg	Single dose	Negative 48 hours after dosing	Mirsalis et al. (1989)
DNA SSBs by alkaline elution	Rat hepatocytes	Inhalation, 3 or 6 hours, 2,000 and 4,000 ppm	3 or 6 hours	Negative at all concentrations and time points	Graves et al. (1994b)
DNA SSBs by alkaline elution	Rat liver homogenate	Gavage, 2 doses, 425 mg/kg and 1,275 mg/kg, administered 4 and 21 hours before liver harvesting	4 or 21 hours (time between dosing and liver harvesting)	Positive at 1,275 mg/kg	Kitchin and Brown (1989)
DNA SSBs by alkaline elution	Rat liver and lung homogenate	Liver: inhalation, 4,000, 5,000 ppm Lung: inhalation, 4,000 ppm	3 hours 3 hours	Negative for both liver and lung at all concentrations	Graves et al. (1995)
DNA-protein cross-links	Hamster liver and lung cells	Inhalation, 6 hr/day, 500, 1,500, 4,000 ppm	3 days	Negative at all concentrations	Casanova et al. (1996)
DNA adducts	Rat liver and kidney cells	Intraperitoneal, 5 mg/kg	Single dose	Negative	Watanabe et al. (2007)

Table 4-34. Comparison of in vivo dichloromethane genotoxicity assays targeted to lung or liver cells, by species

	Studies in mice				Studies in rats			
Assay	**Test system**	**Route, dose (duration)**	**Results**	**Reference**	**Test system**	**Route, dose (duration)**	**Results**	**Reference**
DNA synthesis	Liver	Gavage, 1,000 mg/kg; inhalation, 4,000 ppm (2 hours)	Negative in oral and inhalation studies	Lefevre and Ashby (1989)				No studies
Unscheduled DNA synthesis	Hepatocytes	Inhalation, 2,000 and 4,000 ppm. (2 or 6 hours)	Negative	Trueman and Ashby (1987)	Hepatocytes	Inhalation, 2,000 and 4,000 ppm (2 or 6 hours)	Negative	Trueman and Ashby (1987)
Unscheduled DNA synthesis					Hepatocytes	Intraperitoneal, 400 mg/kg	Negative	Mirsalis et al. (1989)
Sister chromatid exchange	Lung cells	Inhalation 6 hr/day, 5 days/wk, 0, 4,000, 8,000 ppm (2 weeks) Inhalation 6 hr/day, 5 days/wk, 0, 2,000 ppm (12 weeks)	Positive at 8,000 ppm Positive at 2,000 ppm	Allen et al. (1990)				No studies
Chromosome aberrations	Lung cells	Inhalation, 6 hr/day, 5 days/wk, 0, 4,000, 8,000 ppm (2 weeks)	Positive at 8,000 ppm	Allen et al. (1990)				No studies
DNA-protein cross-links	Liver and lung cells	Inhalation, 6 hr/day, 3 days, 4,000 ppm (3 days) Inhalation, 6 hr/day, 150, 500, 1,500, 3,000, 4,000 ppm (3 days)	Positive in liver 4,000 ppm Positive in liver at 500–4,000 ppm; both studies negative in lung	Casanova et al. (1992)				No studies
DNA SSBs by alkaline elution	Hepatocytes	Inhalation, 2,000 and 4,000 ppm (3 or 6 hours)	Positive at 4,000 ppm	Graves et al. (1994b)	Hepatocytes	Inhalation, 3 or 6 hours, 2,000 and 4,000 ppm	Negative at all concentrations and time points	Graves et al. (1994b)
DNA SSBs by alkaline elution	Liver and lung homogenate	Liver: inhalation, 2,000, 4,000, 6,000, 8,000 ppm (3 hours) Lung: inhalation, 1,000, 2,000, 4,000, 6,000 ppm (3 hours)	Liver: Positive at 4,000–8,000 ppm Lung: Positive at 2,000–4,000 ppm	Graves et al. (1995)	Liver and lung homogenate	Liver: inhalation, 4,000, 5,000 ppm Lung: inhalation, 4,000 ppm	Negative in liver and lung at all concentrations and time points	Graves et al. (1995)
DNA SSBs by alkaline elution					Liver homogenate	Gavage, 425 mg/kg and 1,275 mg/kg	Positive at 1,275 mg/kg	Kitchin and Brown (1989)

Table 4-34. Comparison of in vivo dichloromethane genotoxicity assays targeted to lung or liver cells, by species

	Studies in mice				Studies in rats			
Assay	**Test system**	**Route, dose (duration)**	**Results**	**Reference**	**Test system**	**Route, dose (duration)**	**Results**	**Reference**
DNA damage by comet assay	Liver and lung cells	Gavage, 1,720 mg/kg; organs harvested at 0 (control), 3, and 24 hours	Positive only at 24 hours after dosing	Sasaki et al. (1998)				No studies
DNA adducts	Liver and kidney cells	Intraperitoneal, 5 mg/kg	Negative	Watanabe et al. (2007)	Liver and kidney cells	Intraperitoneal, 5 mg/kg	Negative	Watanabe et al. (2007)

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4.5.2. Mechanistic Studies of Liver Effects

One of the major target organs from dichloromethane exposure is the liver, and several studies have focused on examining the potential mechanisms producing liver tumors. This section summarizes the primary mechanistic studies that were conducted in order to examine the hepatic tumors produced by dichloromethane in mice. A parallel set of studies, discussed in the next section, focus on potential mechanisms that produce lung tumors. Briefly, dichloromethane-induced liver tumors first appeared in mice after 52 weeks of exposure (Maronpot et al., 1995; Kari et al., 1993), which was when tumors began to appear in control mice, indicating a similar time course in tumor formation between treated and untreated groups. Onset of liver tumor formation is not preceded by liver cell proliferation (Casanova et al., 1996; Foley et al., 1993). Further mechanistic studies were conducted to assay the tumor for significant changes in proto-oncogene activation and tumor suppressor gene inactivation (Maronpot et al., 1995; Devereux et al., 1993; Hegi et al., 1993). A second subset of mechanistic studies was conducted to elucidate the reason that mice are the most sensitive species to liver tumors and if other species exhibited changes in liver function (Thier et al., 1998; Reitz et al., 1989). It was found that mice have the highest level of GST-T1 catalytic activity but that humans, rats, and hamsters among other species also metabolize dichloromethane in the liver to a GST conjugate. In contrast, there has been little research focusing on the mechanisms through which nonneoplastic hepatic effects (seen most strongly in the rat) are produced, and the role of the parent material, metabolites of the CYP2E1 pathway, metabolites of the GST pathway, or some combination of parent material and metabolites is not known.

Liver tumor characterization studies

Several studies have examined the time course of appearance of liver tumors in B6C3F_1 mice exposed to 2,000 or 4,000 ppm and possible links between hepatic nonneoplastic cytotoxicity, enhanced hepatic cell proliferation, and the development of liver tumors (Casanova et al., 1996; Maronpot et al., 1995; Foley et al., 1993; Kari et al., 1993). The studies provide no clear evidence for a sustained liver cell proliferation response to dichloromethane that can be linked to the development of dichloromethane-induced liver tumors. Additionally, a few studies have examined if dichloromethane-induced liver tumors are the result of proto-oncogene activation and tumor suppressor gene inactivation (Maronpot et al., 1995; Devereux et al., 1993; Hegi et al., 1993).

Kari et al. (1993) (also summarized by Maronpot et al. [1995]) reported data from 6 groups of 68 female B6C3F_1 mice exposed to six ”stop-exposure“ protocols of differing durations and sequences, with each exposure concentration standardized at 2,000 ppm for 6 hours per day, 5 days per week. The six stop-exposure protocols were 26 weeks of exposure followed by 78 weeks without exposure, 78 weeks without exposure followed by 26 weeks of exposure, 52 weeks without exposure followed by 52 weeks with exposure, 52 weeks exposed

followed by 52 weeks without exposure, 78 weeks exposed followed by 26 weeks without exposure, and 26 weeks without exposure followed by 78 weeks with exposure. A control group (no exposure, 104 weeks duration) and a maximum exposure (104 weeks duration) group were also included. Exposure for 26 weeks did not result in an increased incidence of liver tumors (adenomas or carcinomas). Respective percentages of animals with liver tumors were 27% (18/67), 40% (27/67), and 34% (23/67) for the controls, early 26-week exposure and late 26-week exposure groups, respectively. Exposure to 2,000 ppm for 52 weeks (followed by no exposure until 104 weeks), 78 weeks (either early or late exposure periods), or 104 weeks produced increased incidence of mice with liver tumors ($p < 0.05$), but this increase was not seen in the 52-week late exposure group. Respective percentages of animals with liver tumors (adenomas or carcinomas combined) were 44% (28/64), 31% (21/67), 62% (42/68), 48% (32/67) and 69% (47/68) for the 52 (early exposure), 52 (late exposure), 78 (early exposure), 78 (late exposure), and 104 week exposure periods, respectively. With the 78 week exposures, the difference in the liver tumor incidence between the early and late exposure periods was statistically significant ($p < 0.01$). A greater increase in multiplicity of liver tumors was also seen with the early 78-week exposure period. These data suggest that 52 weeks of exposure was required to increase the incidence of liver tumors in mice, that early exposure was more effective than late exposure, and that the increased risk continued after cessation of exposure.

Histopathologic examination of liver tissue at interim killings at eight time periods (13, 26, 52, 68, 75, 78, 83, or 91 weeks) of exposure to 2,000 ppm (n = 20 mice per killing) found no evidence of nonneoplastic cytotoxicity that preceded the appearance of proliferative neoplastic liver lesions. Incidences of mice with liver adenomas or carcinomas were elevated (between 40–60%) at five of the six interim killings after 52 weeks. The incidence rates at each time period were 0/20 (0%) at 13 weeks, 1/20 (5%) at 26 weeks, 8/20 (40%) at 52 weeks, 4/26 (15%) at 68 weeks, 13/20 (65%) at 75 weeks, 12/19 (63%) at 78 weeks, 8/20 (40%) at 83 weeks, and 20/30 (66%) at 91 weeks. The collected liver lesion data identify no exposure-related increased incidence of nonneoplastic liver lesions that could be temporally linked to liver tumor development. Liver tumors first appeared at about the same time in control and exposed animals (52 weeks).

Foley et al. (1993) examined indices of cell proliferation in livers of female $B6C3F_1$ mice exposed to 1,000, 2,000, 4,000, or 8,000 ppm dichloromethane (6 hours/day, 5 days/week) for 1, 2, 3, or 4 weeks or to 2,000 ppm for 13, 26, 52, or 78 weeks but found no evidence for sustained cell proliferation with prolonged exposure to dichloromethane. To label liver cells in S phase, tritiated thymidine (1- to 4-week exposure protocols) or bromodeoxyuridine (13- to 78-week protocols) was administered subcutaneously via an osmotic mini-pump for 6 days prior to killing. Labeled hepatocytes in liver sections (from 10 mice in each exposure/duration group) were counted to assess the number of cells in S-phase per 1,000 cells. S-phase labeling indices in livers of exposed mice at most killings were equivalent to or less than those in control mice.

A transient increase in S-phase labeling index of about two- to fivefold over controls was observed at the 2-week killing of mice exposed to 1,000, 4,000, or 8,000 ppm. Because of the transient nature and small magnitude of the response, it is not expected to be of significance to the promotion of liver tumors in chronically exposed mice. Foley et al. (1993) also compared cell proliferation labeling indices in foci of cellular alteration and non-affected liver regions in control and exposed mice but found no significant difference between control and exposed mice. S-phase labeling was accomplished by immunohistologic staining for proliferating cell nuclear antigen in liver sections from 24 control mice and 15 exposed mice, with livers showing foci of cellular alteration. In both control and exposed livers, the labeling index was about four- to fivefold higher in foci of cellular alteration than in surrounding unaffected liver tissue.

In mice exposed to 2,000 ppm for 13–78 weeks, relative liver weights were statistically significantly elevated compared with controls; about 10% increased at 13 and 26 weeks and about 30–40% increased at 52 and 78 weeks. Histologic changes in liver sections of 2,000 ppm mice exposed for 13–78 weeks were restricted to hepatocellular hypertrophy (observed at all killing intervals) and preneoplastic (foci of cellular alteration) and neoplastic (adenoma and carcinoma) lesions. No signs of liver tissue degeneration were found. Adenoma and focus of alteration were first detected at 26 weeks (2/10 versus 0/10 in controls). At 52 weeks, 4/10 exposed mice had proliferative lesions (1 focus, 1 adenoma, and 2 carcinomas), compared with 1/10 in controls (1 adenoma). At 78 weeks, 7/10 exposed mice had proliferative lesions (2 foci, 3 adenomas, 6 carcinomas) compared with 1/10 in controls (1 adenoma). In summary, the results indicate that inhalation exposure to 2,000 ppm dichloromethane produced an increase incidence of liver tumors in female $B6C3F_1$ mice. No evidence was found for sustained cell proliferation or liver tissue degeneration in response to dichloromethane exposure, but exposure was associated with relative liver weight increases and hepatocellular hypertrophy.

Casanova et al. (1996) found no clear evidence of increased cell proliferation in the livers of male $B6C3F_1$ mice exposed to dichloromethane concentrations >1,500 ppm 6 hours/day for 3 days. Three or four groups of three mice were exposed to 146, 498, 1,553, or 3,923 ppm unlabeled dichloromethane for 2 days and then exposed to [^{14}C]-labeled dichloromethane for 6 hours on the third day. Radiolabel incorporated into liver DNA deoxyribonucleosides was measured as an index of cell proliferation. Radiolabel incorporated into liver DNA deoxyribonucleosides increased approximately fivefold from 146 to 1,553 ppm, but further increases were not apparent at 3,923 ppm. (In contrast, as described in section 4.5.3, radiolabel incorporation into lung DNA deoxyribonucleosides displayed a 27-fold increase over this concentration range.) The small magnitude of the increase in radiolabel incorporation into liver DNA deoxyribonucleosides with increasing exposure concentration suggests that little, if any, enhanced cell proliferation occurred in the liver in response to dichloromethane exposure.

Devereux et al. (1993) (also reported in Maronpot et al. [1995]) analyzed liver tumors in female $B6C3F_1$ mice for the presence of activated H-*ras* oncogenes. Fifty dichloromethane-

induced and 49 spontaneous liver tumors were screened for H-*ras* mutations. There was a relatively high frequency of activated H-*ras* among the nonexposed $B6C3F_1$ mice: 67% of the spontaneous tumors and 76% of the dichloromethane-induced tumors contained mutations in the H-*ras* gene. Overall, the mutation profile of the dichloromethane-induced tumors did not significantly differ from the spontaneous tumors.

Similarly, Hegi et al. (1993) analyzed the liver tumors from female $B6C3F_1$ mice for inactivation of the tumor suppressor genes, *p53* and *Rb-1*. Half of the liver tumors used for this study had an activated H-*ras* oncogene. Twenty liver tumors (15 carcinomas and 5 adenomas) were screened for loss of heterozygosity (LOH) on chromosome 11 and 14, which is associated with malignant conversion of the *p53* gene (chromosome 11) and the *Rb-1* gene (chromosome 14). Only one tumor out of 20 contained a LOH at chromosome 14 and no dichloromethane-induced liver tumors contained a LOH at chromosome 11.

Liver metabolic studies

As described in detail in section 3.3, GST-T1 enzymatic activity and distribution is variable among species, and there is also considerable intraspecies variability among humans. In summary, Reitz et al. (1989) demonstrated a greater metabolic activity with respect to dichloromethane in livers of $B6C3F_1$ mice compare with F344 rats, Syrian golden hamsters, and humans. The rates of in vitro metabolism by the GST pathway were about 4-, 12-, and 20-fold greater in $B6C3F_1$ mouse liver samples than in F344 rat, human, and Syrian golden hamster samples, respectively (Reitz et al., 1989). A more recent study characterized the dichloromethane metabolic capacity specifically of hepatic GST-T1 (Thier et al., 1998). Enzymatic activities of GST-T1 in liver from F344 rats, $B6C3F_1$ mice, Syrian golden hamsters, and humans with three different GST-T1 phenotypes (nonconjugators, low conjugators, high conjugators) showed the following order with dichloromethane as a substrate: mouse >> rat > human high conjugators>human low conjugators > hamster > human nonconjugators.

4.5.3. Mechanistic Studies of Lung Effects

The finding of increased lung tumors in $B6C3F_1$ mice exposed to dichloromethane (Mennear et al., 1988; NTP, 1986) has stimulated a number of studies designed to examine the mechanism for dichloromethane-induced lung tumors in this animal. The lung tumor mechanism studies were conducted with $B6C3F_1$ mice, and the frequency of lung tumors in control animals was very low. Time-course studies for lung tumor development were conducted, and it was found that the onset of lung tumor development was much shorter than liver tumors (Kari et al., 1993) (reported in Maronpot et al. [1995]). As a result, it is hypothesized that a potential common mechanism independent of liver metabolism is producing tumors in the lung. As with the liver tumors, there were no significant increases in mutations for the K-*ras* oncogene (Devereux et al., 1993) or the *p53* and *Rb-1* tumor suppressor genes (Hegi et al., 1993).

Additionally, the Clara cells, which are non-ciliary secretory cells found in the primary bronchioles of the lung, are selectively targeted after dichloromethane exposure. Acute dichloromethane exposure produces Clara cell vacuolization, which is not sustained with long-term exposure (Foster et al., 1992). There is a correlation between the acute effects on the Clara cell and the lung tumors from chronic exposure to dichloromethane (Kari et al., 1993). However, the exact mechanism for producing these lung effects is not completely understood at this point. Provided below is a summary of the studies examining the potential mechanisms for producing lung tumors resulting from dichloromethane exposure.

Lung tumor characterization studies

Kari et al. (1993) (also summarized in Maronpot et al. [1995]) demonstrated that only 26 weeks of exposure to 2,000 ppm was necessary to produce significantly increased incidence of female $B6C3F_1$ mice with lung tumors. In the six "stop-exposure" protocol experiment (26 weeks exposure followed by 78 weeks without exposure, 78 weeks without exposure followed by 26 weeks exposure, 52 weeks without exposure followed by 52 weeks with exposure, 52 weeks exposed followed by 52 weeks without exposure, 78 weeks exposed followed by 26 weeks without exposure, and 26 weeks without exposure followed by 78 weeks with exposure), early but not late exposure for 26 or 52 weeks resulted in an increased incidence of animals with lung tumors (adenoma or carcinomas). Respective percentages of animals with lung tumors were 7.5% (5/67), 31% (21/68), 4% (3/67), 63% (40/63), and 15% (10/67) for the controls, early 26-, late 26-, early 52-, and late 52-week exposure groups, respectively. With the 78-week exposures, both the early and late exposure regimens produced an increased incidence of lung tumors compared with controls (56% [38/68] and 19% [13/68], respectively), compared with the incidence of 63% (42/67) seen in the group exposed for the full 104 weeks. Thus a plateauing of risk was seen, with similar incidence rates seen with the early 52-week, early 78-week, and 104-week exposure periods. The difference in the lung tumor incidence between the early and late exposure periods of similar duration was statistically significant ($p < 0.01$) for the 26-, 52-, and 78-week duration protocols. A greater increase in multiplicity of lung tumors was also seen with the early 78-week exposure period. As with the liver tumor data from the same series of experiments, these data suggest that early exposure was more effective than late exposure and that the increased risk continued after cessation of exposure.

Histopathologic examination of lung tissue from mice killed at 13, 26, 52, 68, 75, 78, 83, or 91 weeks of exposure to 2,000 ppm (n = 20 mice per killing) found no evidence of nonneoplastic cytotoxicity that preceded the appearance of proliferative neoplastic lung lesions. In contrast, incidences of mice with lung adenomas or carcinomas (combined) were elevated at interim killings ≥52 weeks; incidences for the interim killings of mice exposed to 2,000 ppm (6 hours per day, 5 days per week) between 13 and 91 weeks were 0/20 (0%) at 13 weeks, 0/20 (9%) at 26 weeks, 6/20 (30%) at 52 weeks, 6/26 (23%) at 68 weeks, 8/20 (40%) at

75 weeks, 9/19 (47%) at 78 weeks, 10/20 (50%) at 83 weeks, and 14/30 (47%) at 91 weeks, respectively. Lung hyperplasia was found at an increased incidence only at 91 weeks, well after the 26- and 52-week periods that induced increased incidences of mice with lung tumors.

Kanno et al. (1993) found no evidence for histologic changes or increased cell proliferation in lung tissue of female $B6C3F_1$ mice exposed to 2,000 or 8,000 ppm dichloromethane for 1, 2, 3, or 4 weeks, compared with control mice, or in mice exposed to 2,000 ppm for 13 or 26 weeks. Osmotic mini-pumps were used to deliver tritiated thymidine and label cells undergoing replicative DNA synthesis over 6-day periods before killing. Labeled cells undergoing rapid DNA synthesis and cell proliferation were assessed in sections of proximal and terminal bronchioles and alveoli of lungs from groups of 5 mice exposed for 1–4 weeks or 10 mice exposed for 13 or 26 weeks. There were no exposure-related histopathologic or labeling index changes in the alveoli, but lower labeling indices were found in the bronchiolar epithelium of exposed mice compared with controls.

The combined results from the Kari et al. (1993) and Kanno et al. (1993) studies are consistent with the hypothesis that dichloromethane-induced lung tumors in $B6C3F_1$ mice are not preceded by overt cytotoxicity, enhanced and sustained cell proliferation, or hyperplasia in the lung. Two other studies (Casanova et al., 1996; Foster et al., 1992), however, have reported evidence for enhanced cell proliferation in lungs of $B6C3F_1$ mice exposed for acute durations to airborne dichloromethane. Only one of these studies (Foster et al., 1992), however, looked for sustained cell proliferation in the lung with prolonged exposure. In agreement with the results from Kanno et al. (1993), no evidence was found for sustained cell proliferation in lungs with prolonged exposure to dichloromethane at concentrations demonstrated to induce lung tumors in mice.

Casanova et al. (1996) detected evidence of increased cell proliferation in the lungs of male $B6C3F_1$ mice exposed to dichloromethane concentrations >1,500 ppm 6 hours/day for 3 days. Three or four groups of three mice were exposed to 146, 498, 1,553, or 3,923 ppm unlabeled dichloromethane for 2 days and then exposed to [^{14}C]-labeled dichloromethane for 6 hours on the third day. Radiolabel incorporated into lung DNA deoxyribonucleosides (after removal of DNA-protein cross links containing radiolabeled formaldehyde) was measured as an index of cell proliferation. Radiolabel incorporation into lung DNA deoxyribonucleosides increased with increasing exposure concentration, with the amount increasing by about 27-fold between 146 and 3,923 ppm. In hamsters that did not develop tumors in response to chronic inhalation exposure to 3,500 ppm dichloromethane (Burek et al., 1984), no evidence for enhanced radiolabel incorporation into lung DNA deoxyribonucleosides was found following acute exposure (Casanova et al., 1996).

Devereux et al. (1993) (also summarized in Maronpot et al. [1995]) analyzed lung tumors in female $B6C3F_1$ mice for the presence of activated K-*ras* oncogenes. Fifty-four dichloromethane-induced and 17 spontaneous lung tumors (7 from the NTP [1986] study and 10

from a study in C57BL/6 × C34F1 mice reported by Candrian et al. [1991]) were screened for K-*ras* mutations. Twenty percent of the dichloromethane-induced tumors and 24% of the spontaneous tumors contained mutations in the K-*ras* gene. Devereux et al. (1993) stated that there may be a significant difference in the incidence of K-*ras* activation between spontaneous and dichloromethane-induced tumors. However, the small number of the spontaneous tumors that were available for the study limits the conclusions that can be made from the results.

Hegi et al. (1993) analyzed the lung tumors from female $B6C3F_1$ mice for inactivation of the tumor suppressor genes, *p53* and *Rb-1*. Forty-nine dichloromethane-induced lung carcinomas, five lung adenomas, and seven spontaneous lung carcinomas were screened for LOH on mouse chromosome 11 and 14, which is associated with malignant conversion of the *p53* gene (chromosome 11) and the *Rb-1* gene (chromosome 14). Fourteen percent (n = 7) of the dichloromethane-induced lung carcinomas exhibited LOH at chromosome 11. No *p53* mutations were detected in the seven spontaneous lung tumors or the five dichloromethane-induced lung adenomas. Only three dichloromethane-induced tumors exhibited LOH at chromosome 14. The authors noted that inactivation of the *p53* and *Rb-1* tumor suppressor genes infrequently occur in lung and liver tumors.

Clara cell studies

Foster et al. (1992) found enhanced cell proliferation in bronchiolar cells and, to a lesser degree, alveolar cells in the lungs of male $B6C3F_1$ mice exposed for acute durations (2, 5, 8, or 9 days) to 4,000 ppm dichloromethane (6 hours/day, 5 days/week), but the response was less distinct after subchronic durations of exposure (89, 92, or 93 days). To measure cell proliferation, mice (n = 5 per exposure-duration group) were given subcutaneous doses of tritiated thymidine for five consecutive days prior to killing. Labeled cells in bronchiolar or alveolar epithelium in lung sections were counted to assess the number of cells in S phase per 1,000 cells. Counts of bronchiolar epithelium cells in S phase in exposed mice sacrificed on days 2, 5, 8, and 9 were approximately 2-, 15-, 3-, and 5-fold higher, respectively, than those of unexposed mice at day 0 of the experiment. In exposed mice sacrificed on days 89, 92, and 93, less than twofold increases in bronchiolar epithelium cell labeling were observed. Increased cell labeling was found in alveolar epithelium only on day 8 (about seven- to eightfold increase) and day 9 (about fourfold increase). Vacuolation of the Clara cells of the bronchiolar epithelium was observed on day 2 (scored as ++, majority of cells affected), day 9 (+++, virtually all the cells affected), and day 44 (+, moderate effect to cells) but was not apparent on days 5, 8, 40, 43, 89, 92, or 93. No hyperplasia of the bronchiolar epithelium or changes to Type II alveolar epithelial cells were observed in the lungs of any of the exposed mice at any time point. The appearance and disappearance of the Clara cell vacuolation were generally correlated with the appearance and disappearance of enhanced cell proliferation in the bronchiolar epithelium; enhanced cell proliferation was observed on days 2, 5, 8, and 9 (along with appearance of Clara cell

vacuolation on days 2 and 9) but was not observed on days 89, 92, and 93 when Clara cell lesions also were not observed. This suggests that cell proliferation was enhanced in response to Clara cell damage but was not sustained with repeated exposure to dichloromethane.

Currently, a mechanistic connection has not been established between the acute effects of dichloromethane on Clara cells in the bronchiolar epithelium and the development of lung tumors in $B6C3F_1$ mice exposed by inhalation to concentrations ≥2,000 ppm dichloromethane for 2 years (NTP, 1986) or for 26 weeks followed by no exposure through 2 years (Maronpot et al., 1995; Kari et al., 1993). There appears to be a concordance between exposure concentrations inducing acute Clara cell vacuolation (≥2,000 ppm) and those inducing lung tumors (≥2,000 ppm). However, transient acute Clara cell vacuolation does not appear to progress to necrosis or lead to sustained cell proliferation (which could promote the growth of tumor-initiated cells) and appears to be dependent on CYP metabolism of dichloromethane (see the following paragraphs discussing pertinent findings reported by Foster et al. [1994, 1992]). In contrast, there is consistent and specific evidence for an association between the formation of DNA-reactive GST-pathway metabolites and the formation of lung and liver tumors from inhalation exposure (see sections 4.5.2 and 4.7.3).

Foster et al. (1992) noted that the appearance and disappearance of Clara cell vacuolation in mouse lungs showed concordance with temporal patterns for immunologic staining for CYP2B1 and 2B2 levels in lung sections. A similar temporal pattern was reported for CYP2B1 and 2B2 monooxygenase activities (ethoxycoumarin O-dealkylation or aldrin epoxidation) assayed in lung microsomes. When there was a marked decrease in CYP2B1 and 2B2 staining (e.g., on day 5) or monooxygenase activities, the lesion was not present. Similarly, the appearance of the lesion was preceded (the day before) by the recovery of monooxygenase activities or immunologic staining close to control levels. These patterns suggested to Foster et al. (1992) that Clara cells may have developed tolerance to dichloromethane due to inactivation of a CYP isozyme.

In subsequent studies, increased percentages of vacuolated bronchiolar epithelium cells were noted in mice exposed to 2,000 ppm (26.3 ± 6.7%) or 4,000 ppm (64.8 ± 12.8%), but vacuolated cells were not observed in bronchiolar epithelium of lung sections from control mice or mice exposed to 125, 250, 500, or 1,000 ppm (Foster et al., 1994). Pretreatment with the CYP inhibitor, piperonyl butoxide, counteracted the 2,000 ppm effect (2.4 ± 3.6% vacuolated cells), whereas GSH-depleted mice showed no statistically significant change in percentage of vacuolated cells (32.7 ± 16.9%) compared with the mean percentage in mice exposed to 2,000 ppm without pretreatment. No consistent, statistically significant, exposure-related changes were found in cytosolic GST metabolic activities (with dichloromethane as substrate) or microsomal CYP monooxygenase activities (ethoxycoumarin O-dealkylation), but mean cytosolic levels of nonprotein sulfhydryl compounds were elevated in lungs of mice exposed to 1,000 and 2,000 ppm (134.6 ± 17.1 and 146.4 ± 6.7 nmol/mg protein, respectively) compared

with control levels (109.5 ± 7.6 nmol/mg protein). Increased cell proliferation was found in cultured Clara cells isolated from 4,000 ppm exposed mice compared with nonexposed mice; respective values for percentage of cells in S phase were 18.97 ± 1.18 and 2.02 ± 0.86% (Foster et al., 1994).

Results from the studies by Foster et al. (1994, 1992) indicate that 6-hour exposures of B6C3F$_1$ mice to dichloromethane concentrations ≥2,000 ppm caused transient Clara cell vacuolation in the bronchiolar epithelium, which was not consistently observed following repeated exposures. With repeated exposure to 4,000 ppm, the Clara cell vacuolation did not progress to necrosis, and no hyperplasia of the bronchiolar epithelium was found after up to 13 weeks of exposure. The transient Clara cell vacuolation was decreased by CYP inhibition with piperonyl butoxide and was unaffected by GSH depletion, indicating that a CYP metabolite was involved. Clara cell vacuolation was not found after five consecutive daily 6-hour exposures to 4,000 ppm but reappeared after 2 days without exposure followed by two additional consecutive daily exposures (day 9). With repeated exposure, the lesion was detected at a diminished severity on day 44 (but was not found on day 40 or 43) and on day 93 (but was not found on day 89 or 92). The temporal pattern of Clara cell vacuolation with repeated exposure was reflected in the occurrence of transiently decreased CYP metabolic activity after the appearance of vacuolation. Foster et al. (1994, 1992) proposed that the diminishment of severity, or the disappearance, of the Clara cell vacuolation with repeated exposure was due to the development of a tolerance to dichloromethane, linked with a decrease of CYP metabolism of dichloromethane.

4.5.4. Mechanistic Studies of Neurological Effects

Several neurobehavioral studies (see section 4.4.3 for a complete summary) have demonstrated that dichloromethane exposure results in decreased spontaneous motor activity with pronounced lethargy at high concentrations (1,000 ppm or greater). These effects, combined with the observation that dichloromethane impairs learning and memory (Alexeef and Kilgore, 1983) and affects production of evoked responses to sensory stimuli (Herr and Boyes, 1997; Rebert et al., 1989), indicate that dichloromethane produces significant neurological effects. The mechanisms behind producing these changes have been examined by measuring changes in neurotransmitter levels and changes in neurotransmitter localization. Specific brain regions (e.g., hippocampus, caudate nucleus, cerebellum) were analyzed to determine if dichloromethane-induced behavioral effects, such as learning and memory (hippocampus, caudate nucleus) and movement (cerebellum), are resulting from pathological changes in these regions. Changes in neurotransmitter levels were also monitored to see if there was any correlation in behavior and neurochemical changes. Summaries of these studies are provided below. It is not yet known if dichloromethane directly interacts with neuronal receptors, as has been demonstrated for toluene and ethanol, two other solvents with neurobehavioral and

neurophysiological profiles that are similar to those of dichloromethane (for a review see Bowen et al. [2006]).

Kanada et al. (1994) examined the effect of dichloromethane on acetylcholine and catecholamines (dopamine, norepinephrine, serotonin) and their metabolites in the midbrain, hypothalamus, hippocampus, and medulla from male Sprague-Dawley rats (four to five per group). The rats were sacrificed 2 hours after a single gavage dose of 0 or 534 mg/kg of undiluted dichloromethane. Administration of dichloromethane significantly increased the concentration of acetylcholine in the hippocampus and increased dopamine and serotonin levels in the medulla. Dichloromethane decreased norepinephrine levels in the midbrain, and hypothalamus and serotonin levels were decreased in the hypothalamus. There was a trend toward decreased dopamine in the hypothalamus, but the variability between the animals was so high that the effect was not significant. The authors speculated that increased acetylcholine release from dichloromethane administration may be due to decreased acetylcholine release from the nerve terminals. It is unclear as to how these neurochemical changes could be correlated to the neurobehavioral changes observed after dichloromethane exposure.

In a 2-week exposure study, male Wistar rats were exposed to dichloromethane at 500 or 1,000 ppm (6 hours/day, 5 days/week for 1 or 2 weeks) or 1,000 ppm TWA (1 hour at 100 ppm, 1 hour peak at 2,800 ppm, 1 hour at 100 ppm, repeated immediately, 5 days/week for 1 or 2 weeks) (Savolainen et al., 1981). Brains were removed from rats at the end of the study and analyzed. The 1,000 ppm TWA group displayed increases in cerebral RNA. Other changes noted for this group in the cerebrum included significant increases in NADPH-diaphorase and succinate dehydrogenase activity. These changes suggest increased neural activity to possibly offset the overall inhibitory effect of dichloromethane in the CNS. It could also possibly explain why lethargic effects decrease with continued dichloromethane exposure, and this result demonstrates a neuroprotective mechanism resulting from dichloromethane exposure. After a 7-day withdrawal, RNA levels in the cerebrum were significantly lower in the 1,000 ppm group. Succinate dehydrogenase levels remained lowered in the 1,000 ppm TWA group after the 7-day exposure-free period.

Changes in brain catecholamine levels after a subacute exposure to dichloromethane were evaluated using male Sprague-Dawley rats (Fuxe et al., 1984). Rats were exposed to 70, 300, and 1,000 ppm dichloromethane, 6 hours/day for 3 consecutive days. At all exposures, there was a significant decrease of catecholamine concentrations in the posterior periventricular region of the hypothalamus. The impact of dichloromethane was also evaluated on the hypothalamic-pituitary gonadal axis. The hypothalamus regulates secretion of reproductive hormones, such as follicle-stimulating hormone and luteinizing hormone. The levels of the hormone release were not significantly changed with dichloromethane exposure. In the caudate nucleus, which is involved in memory processes, the catecholamine level initially increased (at 70 ppm) and then was lower (1,000 ppm) in comparison to the control. The study overall demonstrates significant

changes in catecholamine levels in the hypothalamus and caudate nucleus. Catecholamine level changes in the hypothalamus did not have any significant effect on hormonal release and decreased catecholamine levels in the caudate nucleus at higher exposures may lead to memory and learning impairment.

A series of studies were conducted in male and female Mongolian gerbils exposed continuously to ≥210 ppm dichloromethane for 3 months, followed by a 4-month exposure-free period (Karlsson et al., 1987; Briving et al., 1986; Rosengren et al., 1986). Decreased DNA concentrations were noted in the hippocampus at both the 210 and 350 ppm exposures. At 350 ppm, there was also decreased DNA concentration in the cerebellar hemispheres, indicating a decreased cell density in these regions probably due to cell loss (Rosengren et al., 1986). These findings indicate that the cerebellum, which is the section of the brain that regulates motor control, is a target for dichloromethane. In the same study, increased astroglial proteins were found in the frontal and sensory motor cerebral cortex, which directly correlated to the astrogliosis that was observed in those areas. Up-regulation of these astroglial proteins is a good indicator of neuronal injury (Rosengren et al., 1986).

Karlsson et al. (1987) measured DNA concentrations in different regions of the gerbil brain. The total brain protein concentration per wet weight was not significantly different between dichloromethane-exposed and control animals. However, DNA concentrations per wet weight were significantly decreased in the hippocampus after dichloromethane exposure. No other examined regions demonstrated significant changes in DNA concentrations after dichloromethane exposure. Therefore, this result indicates that the hippocampus, which plays a role in the formation of new memories, is another target for dichloromethane in the CNS. This selective DNA concentration decrease observed in the hippocampus is a sign of neurotoxicity as noted by the authors and may possibly explain why some studies have noted memory and learning deficits with dichloromethane exposure.

At a 210 ppm exposure, Briving et al. (1986) observed that dichloromethane decreased glutamate, γ-aminobutyric acid, and phosphoethanolamine levels in the frontal cortex, while glutamate and γ-aminobutyric acid were increased in the posterior cerebellar vermis. Increased levels of glutamate in the posterior cerebellar vermis could reflect an activation of astrocytic glia, since glutamine synthetase is localized exclusively in astrocytes.

4.6. SYNTHESIS OF MAJOR NONCANCER EFFECTS

4.6.1. Oral Exposures

4.6.1.1. *Summary of Human Data*

Information on noncancer effects in humans exposed orally to dichloromethane are restricted to case reports of neurological impairment, liver and kidney effects (as severe as organ failure), and gastrointestinal irritation in individuals who ingested amounts ranging from about 25 to 300 mL (Chang et al., 1999; Hughes and Tracey, 1993). Neurological effects with these

individuals consisted of general CNS depressive symptoms, such as drowsiness, confusion, headache, and dizziness. Hemoglobinuria has been noted as a kidney effect associated with ingestions.

4.6.1.2. *Summary of Animal Data*

Acute oral or intraperitoneal administration of dichloromethane in animals has resulted in several significant effects. General activity and function were affected as evidenced by decreased neuromuscular activity (Moser et al., 1995). Additionally, decreased sensorimotor function was detected through measurement of evoked potentials (Herr and Boyes, 1997) and by using the FOB (Moser et al., 1995). Neurochemical changes (e.g. acetylcholine, dopamine, norepinephrine, serotonin) were detected 2 hours after oral dosage of dichloromethane within specific parts of the brain. It should be noted that all the acute effects that were observed after oral or intraperitoneal administration occurred within 5 hours after dosage. No other significant organ effects were noted after a single acute oral exposure, but in oral pharmacokinetic studies it is known that dichloromethane is primarily distributed to the liver, lungs, and kidneys (Angelo et al., 1986a).

Results from short-term, subchronic, and chronic oral toxicity studies in laboratory animals are summarized in Table 4-35. The data indicate that rats may be more sensitive than mice to nonneoplastic liver effects from orally administered dichloromethane, as evidenced by lower NOAELs and LOAELs, with more severe liver effects in rats. The most frequently observed liver effect was hepatocyte vacuolation, seen with drinking water exposure (90 days) in F344 rats at ≥166 mg/kg-day and B6C3F_1 mice at 586 mg/kg-day (Kirschman et al., 1986) and with gavage exposure (14 days) in CD-1 mice at 333 mg/kg-day (Condie et al., 1983). Hepatocyte degeneration or necrosis was observed in female F344 rats exposed in drinking water for 90 days to 1,469 mg/kg-day (Kirschman et al., 1986) and in female F344 rats exposed by gavage for 14 days to 337 mg/kg-day (Berman et al., 1995), but was not seen in a 90-day drinking water study in B6C3F_1 mice exposed to doses as high as 2,030 mg/kg-day (Kirschman et al., 1986). In the chronic-duration (2-year) study, liver effects were described as nonneoplastic foci and areas of alteration in F344 rats exposed to drinking water doses between 50 and 250 mg/kg-day; an increased incidence of fatty changes in the liver was also noted but the incidence of the latter was not provided (Serota et al., 1986a). These effects were considered to be nonneoplastic for several reasons. Serota et al., (1986b) observed a dose-related increased incidence of 0, 65, 92, 97, 98 and 100% in male rats and 51, 41, 73, 89, 91 and 85% in female rats for the 0, 5, 50, 125, 250 and 250 with recovery groups, respectively. Evidence for liver tumors has been reported in female rats only. Specifically, evidence for liver tumors in rats includes a small number of hepatocellular carcinomas observed in female rats at 50 and 250 mg/kg-day, which reached statistical significance (for trend and for individual pairwise comparisons) only with the combined grouping of neoplastic nodules and hepatocellular

carcinomas. In male rats, only one hepatocellular carcinoma was observed in all of the exposure groups (compared to 4 in the controls), and the incidence of neoplastic nodules and hepatocellular carcinomas was higher in controls (16%) than in any exposure group (16, 3, 0, 6, 5 and 13% for the 0, 5, 50,125, 250 mg/kg-day and 250 with recovery groups, respectively). The authors (Serota et al., 1986a) did not elaborate on the characterization of the altered foci. However, the characterization of altered foci could range from a focal change in fat distribution (nonneoplastic effect) to enzyme altered foci which are generally considered a precursor to tumor formation (Goodman et al., 1994). Serota et al (1986a) reported an increased incidence of fatty change in the liver at doses of 50 mg/kg-day and higher, but the incidence was not reported. In addition, a 90-day study (Kirschman et al., 1986) demonstrated that increased fatty deposits were present in the hepatocyte vacuoles. Therefore, the altered foci (i.e. change in fat distribution) observed by Serota et al., (1986b) may represent a precursor to fatty liver changes which is considered a nonneoplastic effect. Taken together, the data support the conclusion that the altered foci were nonneoplastic.

The NOAEL and LOAEL, 101 and 337 mg/kg-day, for altered neurological functions in female F344 rats (as reported by Moser et al. [1995]) were identical to those reported by Berman et al. (1995) for hepatocyte necrosis in female F344 rats. In the 90-day (Kirschman et al., 1986) and 104-week (Serota et al., 1986a, b) drinking water studies, no obvious clinical signs of neurological impairment were observed in rats or mice at exposure levels that induced liver effects (see Table 4-35), but this study did not include a standardized neurological testing battery.

Table 4-35. NOAELs and LOAELs in selected animal studies involving oral exposure to dichloromethane for short-term, subchronic, or chronic durations

Type of effect and exposure, reference	Species and exposure details	Results	NOAEL (mg/kg-day)	LOAEL (mg/kg-day)
Hepatic, 14-day gavage				
Berman et al. (1995)	F344 rat, female, 8/dose group 0, 34, 101, 337, 1,012 mg/kg-day	Hepatocyte necrosis	101	337
Condie et al. (1983)	CD-1 mouse, male, 5/group for histological examinations; 8/group for blood urea nitrogen, serum creatinine, and serum glutamate-pyruvate transaminase; 0, 133, 333, 665 mg/kg-day	Hepatocyte vacuolation (minimal to mild in 3/5)	133	333
Hepatic, 90-day drinking water				
Kirschman et al. (1986)	F344 rat, male and female; 15/sex/group; males 0, 166, 420, 1,200 mg/kg-day females 0, 209, 607, 1,469 mg/kg-day	Hepatic vacuolation (generalized, centrilobular, or periportal, at lowest dose, in 10/15 males and 13/15 females compared with 1/15 males and 6/15 females in controls)	Not identified	166
Kirschman et al. (1986)	$B6C3F_1$ mouse, male and female, males 0, 226, 587, 1,911 mg/kg-day females 0, 231, 586, 2,030 mg/kg-day	Hepatic vacuolation (increased severity of centrilobular fatty change in mid- and high-dose groups compared with controls)	231	586
Hepatic, 104-week drinking water				
Serota et al. (1986a)	F344 rat, male and female, 0, 5, 50, 125, 250 mg/kg-day	Liver foci/areas of alteration (considered nonneoplastic histologic changes); fatty liver changes also seen at same doses but incidence data not reported; no evidence that increased altered foci progresses to liver tumor formation	5	50
Serota et al. (1986b) Hazelton Laboratories (1983)	$B6C3F_1$ mouse, male and female, 0, 60, 125, 185, 250 mg/kg-day	Some evidence of fatty liver; marginal increase in the Oil Red-O-positive material in the liver	185	250
Neurologic, 14 day				
Moser et al. (1995)	F344 rat, female, 0, 34, 101, 337, 1,012 mg/kg-day	FOB 24 hours postexposure: altered autonomic, neuromuscular, and sensorimotor and excitability measures	101	337
Reproductive				
General Electric Co. (1976)	Charles River CD rat, male and female, gavage for 90 d before mating (10 days between last exposure and mating period); 0, 25, 75, 225 mg/kg-day; F1 offspring received same treatment as parents for 90 days	Reproductive performance of F0 and histologic examination of tissues from F1 offspring	225	Not identified

Table 4-35. NOAELs and LOAELs in selected animal studies involving oral exposure to dichloromethane for short-term, subchronic, or chronic durations

Type of effect and exposure, reference	Species and exposure details	Results	NOAEL (mg/kg-day)	LOAEL (mg/kg-day)
Raje et al. (1988)	Swiss-Webster mouse, male, 0, 250, 500 mg/kg (subcutaneous injection), 3× per week, 4 weeks prior to mating with nonexposed females (1 week between last exposure and mating period)	No statistically significant effects on testes, number of litters, live fetuses/litter, percent dead fetuses/litter, percent resorbed/litter, or fertility index	200	Not identified
Developmental				
Narotsky and Kavlock (1995)	F344 rat, pregnant female, gavage on GDs 6–19; 0, 338, 450 mg/kg-day	Maternal: weight gain depression	338	450
		Fetal: no effects on pup survival, resorptions, pup weight	450	Not identified
Other developmental				
No studies				

Results from a limited number of studies do not provide evidence for effects on reproductive or developmental endpoints (Table 4-35). No effects on pup survival, resorptions, or pup weight were found following exposure of pregnant F344 rats to doses as high as 450 mg/kg-day on GDs 6–19, a dose that depressed maternal weight gain (Narotsky and Kavlock, 1995), and no effects on reproductive performance endpoints (fertility index, number of pups per litter, pup survival) were found in Charles River CD rats exposed for 90 days before mating to doses as high as 225 mg/kg-day. There are no oral exposure studies focusing on neurobehavioral effects or other developmental outcomes.

4.6.2. Inhalation Exposures

4.6.2.1. *Summary of Human Data*

As discussed in sections 4.1.3.1 and 4.1.3.2, acute inhalation exposure of humans to dichloromethane has been associated with cardiovascular impairments due to decreased oxygen availability from COHb formation and neurological impairment from interaction of dichloromethane with nervous system membranes. Results from studies of acutely exposed human subjects indicate that acute neurobehavioral deficits, measured, for example, by psychomotor tasks, tests of hand-eye coordination, visual evoked response changes, and auditory vigilance, may occur at concentrations >200 ppm with 4–8 hours of exposure (Bos et al., 2006; American Conference of Governmental Industrial Hygienists [ACGIH], 2001; ATSDR, 2000; Cherry et al., 1983; Putz et al., 1979; Gamberale et al., 1975; Winneke, 1974).

The clinical and workplace studies of noncancer health effects of chronic dichloromethane exposure have examined markers of disease and specific clinical endpoints relating to cardiac, neurological disease, hepatic function, and reproductive health. As summarized in section 4.1.2.9, the limited available data do not provide evidence of cardiac damage related to dichloromethane exposure in occupationally exposed workers (Hearne and Pifer, 1999; Tomenson et al., 1997; Gibbs et al., 1996; Lanes et al., 1993; Ott et al., 1983d; Cherry et al., 1981). Relatively little is known about the long-term neurological effects of chronic exposures, although there are studies that provide some evidence of an increased prevalence of neurological symptoms among workers with average exposures of 75–100 ppm (Cherry et al., 1981), long-term effects on some neurological measures (i.e., possible detriments in attention and reaction time in complex tasks) in workers whose past exposures were in the 100–200 ppm range (Lash et al., 1991), and an increased risk of suicide in worker cohort studies (Hearne and Pifer, 1999; Gibbs, 1992). Given the suggestions from these studies and their limitations (particularly with respect to sample size and power considerations), the statement that there are no long-term neurological effects of chronic exposures to dichloromethane cannot be made with confidence. With respect to markers of hepatic damage, three studies measured serum enzyme and bilirubin levels in workers exposed to dichloromethane (Soden, 1993; Kolodner et al., 1990; Ott et al., 1983c). There is some evidence of increasing levels of serum

bilirubin with increasing dichloromethane exposure (Kolodner et al., 1990; Ott et al., 1983c), but there are no consistent patterns with respect to the other hepatic enzymes examined (serum γ-glutamyl transferase, serum AST, serum ALT). Thus these studies do not provide clear evidence of hepatic damage in dichloromethane exposed workers to the extent that this damage could be detected by these serologic measures.

Only limited, and somewhat indirect, evidence pertaining to immune-related effects of dichloromethane in humans is available. No risk of the broad category of infection- and parasite-related mortality was reported by Hearne and Pifer (1999), but there was some evidence of an increased risk of influenza and pneumonia-related mortality at two cellulose triacetate fiber production work sites in Maryland and South Carolina (Gibbs, 1992).

Few studies have been conducted pertaining to reproductive effects (i.e., spontaneous abortion, low birth weight, or oligospermia) of dichloromethane exposure from workplace settings (Wells et al., 1989; Kelly, 1988; Taskinen et al., 1986) or environmental settings (Bell et al., 1991). Of these, the data pertaining to spontaneous abortion provide the strongest evidence of an adverse effect of dichloromethane exposure. The limitations of the only study (Taskinen et al., 1986) pertaining to this outcome, however, do not allow firm conclusions to be made regarding dichloromethane and risk of spontaneous abortion in humans.

4.6.2.2. *Summary of Animal Studies*

Acute and short-term (up to 7 days) inhalational exposure to dichloromethane has resulted in neurological and hepatocellular changes. Several neurological-mediated parameters were reported, including decreased activity (Kjellstrand et al., 1985; Weinstein et al., 1972; Heppel and Neal, 1944), impairment of learning and memory (Alexeef and Kilgore, 1983), and changes in responses to sensory stimuli (Rebert et al., 1989). Although learning and memory properties were impaired in one acute exposure (47,000 ppm until loss of righting reflex), it should be noted that this effect has not been characterized by using other learning and memory tasks nor any other exposure paradigms. In a 3-day exposure to dichloromethane (70, 300, or 1,000 ppm 6 hours/day), it was found that in the rat brain there were changes in catecholamine (dopamine, serotonin, norepinephrine) in the hypothalamus and caudate nucleus (Fuxe et al., 1984). The catecholamine level changes did not affect hormonal release, which is a primary function of the hypothalamus.

Another acute exposure study examined immunological response as measured by increased streptoccocal pneumonia-related mortality and decreased bactericidal activity of pulmonary macrophages in CD-1 mice following a single 3-hour exposure to dichloromethane at 100 ppm (Aranyi et al., 1986). No effects were seen at 50 ppm. A 4-week inhalation exposure to 5,000 ppm dichloromethane in rats did not result in changes in immune response as measured by the sheep red blood cell assay (Warbrick et al., 2003). These studies suggest a localized, portal-of-entry effect within the lung, without evidence of systemic immunosuppression.

Mouse hepatocytes showed balloon degeneration (dissociation of polyribosomes and swelling of rough endoplasmic reticulum) within 12 hours of exposure to 5,000 ppm (Weinstein et al., 1972). A subacute exposure in Wistar rats to 500 ppm dichloromethane 6 hours/day for 6 days resulted in increased hemochrome content in liver microsomal CYP (Savolainen et al., 1977).

Results pertaining to liver, lung, and neurological effects from longer (>7 days) subchronic and chronic inhalation toxicity studies in laboratory animals are summarized in Table 4-36; reproductive and developmental studies are summarized in Table 4-37.

Table 4-36. NOAELs and LOAELs in animal studies involving inhalation exposure to dichloromethane for subchronic or chronic durations, hepatic, pulmonary, and neurologic effects

Type of effect and exposure period, reference	Species and exposure details	Results	NOAEL	LOAEL
			ppm	
Hepatic, subchronic (13–14 weeks)				
Haun et al. (1971)	Beagle, female (n = 8); rhesus monkeys, females (n = 4); Sprague-Dawley rats, male (n = 20), ICR mice, females (n = 380) 0, 1,000, 5,000 ppm (continuous exposure; 14 weeks)	Fatty liver at 1,000 ppm in dogs, "borderline" liver changes in monkey at 5,000 ppm, mottled liver changes in rats at 5,000 ppm Decreased movement and lethargy at 1,000 ppm in dogs, mice, and monkey.	Not identified Not identified 1,000 Not identified	1,000 dog 1,000 monkey 5,000 rat 1,000 mouse
Haun et al. (1972)	Beagle (n = 16); Rhesus monkey (n = 4); Sprague-Dawley rats (n = 20), ICR mice (n = 20) 0, 25, 100 ppm (continuous exposure; 14 weeks)	CYP levels decreased in liver microsomes in mice at 100 ppm Increased fatty liver content at 25, 100 ppm in rats	25; Not identified Not identified 25	100 dog 25 monkey 25 rat 100 mouse
Leuscher et al. (1984)	Sprague-Dawley rat, male and female, (20/sex/group) - 0, 1,000 ppm (6 hours/day; 90 days) Beagle, male and female (3/sex/group) - 0, 5,000 ppm	No liver effects noted	1,000 5,000	Not identified Not identified
NTP (1986)	F344/N rat, male and female (10/sex/group) 0, 525, 1,050, 2,100, 4,200, 8,400 ppm (6 hours/day, 5 days/week, 13 weeks)	Decreased lipid:liver weight ratios at 4,200 (females); 8,400 (males); decreased BW by 23% and 11% in males and females at 8,400 ppm compared with controls; one male and one female died at 8,400 ppm before the end of the study.	4,200	8,400
NTP (1986)	$B6C3F_1$ mouse, male and female (10/sex/group) 0, 525, 1,050, 2,100, 4,200, 8,400 ppm (6 hours/day, 5 days/week, 13 weeks)	Hepatocyte centrilobular degeneration at 4,200 females) and 8,400 (males); decreased lipid:liver weight ratios at 8,400 (females); at 8,400 ppm, 4/10 males and 2/10 females died before end of study.	2,100	4,200
Hepatic, 2 years (6 hours/day, 5 day/week)				
Mennear et al. (1988); NTP (1986)	F344/N rat, male and female 0, 1,000, 2,000, 4,000 ppm	Hepatocyte vacuolation and necrosis Hemosiderosis in liver Renal tubular degeneration	Not identified Not identified 1,000	1,000 1,000 2,000

Table 4-36. NOAELs and LOAELs in animal studies involving inhalation exposure to dichloromethane for subchronic or chronic durations, hepatic, pulmonary, and neurologic effects

Type of effect and exposure period, reference	Species and exposure details	Results	NOAEL	LOAEL
			ppm	
Mennear et al. (1988); NTP (1986)	$B6C3F_1$ mouse, male and female 0, 2,000, 4,000 ppm	Hepatocyte degeneration Renal tubule casts	Not identified Not identified	2,000 2,000
Burek et al. (1984)	Syrian golden hamster, male and female 0, 500, 1,500, 3,500 ppm	No effects on histologic, clinical chemistry, urinalytic, and hematologic variables no obvious clinical signs of toxicity	3,500	Not identified
Burek et al. (1984)	Sprague-Dawley rat, male and female 0, 500, 1,500, 3,500 ppm	Hepatocyte vacuolation (M and F) Hepatocyte necrosis (M only), no obvious clinical signs of toxicity)	Not identified 500	500 1,500
Nitschke et al. (1988a)	Sprague-Dawley rat, male and female 0, 50, 200, 500 ppm	Hepatocyte vacuolation significantly increased in females; non-significant increase in males at 500 ppm (31% in controls and 40% in 500 ppm group).	200	500
Pulmonary, 13 weeks (6 hours/day, 5 days/week)				
NTP (1986)	F344 rat, male and female 0, 525, 1,050, 2,100, 4,200, 8,400 ppm	Foreign body pneumonia	4,200	8,400
Foster et al. (1992)	$B6C3F_1$ mouse, male and female 0, 4,000 ppm	Clara cell vacuolation	Not identified	4,000
Neurological, 14 days				
Savolainen et al. (1981)	Wistar rats, male 500, 1,000, 1,000 TWA (100 + 2,800 1-hour peaks[a]) ppm (6 hours/day, 5 days/week, 2 weeks)	Increased RNA in cerebrum at 1,000 ppm; increased enzymatic activities[b] in cerebrum and cerebellum at 1,000 ppm TWA	500	1,000 for brain RNA concentration; 1,000 TWA for brain enzymatic activity
Neurological, 13–14 weeks				
Mattsson et al. (1990)	F344 rat, male and female 0, 50, 200, 2,000 ppm (6 hours/day, 5 days/week)	No exposure-related effects on an observational battery, hind-limb grip strength, a battery of evoked potentials, or histology of brain, spinal cord, peripheral nerves; measured 64 hours postexposure	2,000	Not identified
Haun et al. (1971)	Beagle dogs (female); Rhesus monkeys (female); Sprague-Dawley rats (male); ICR mice (females) 0, 1,000, 5,000 ppm (continuous exposure)	CNS depression most evident in dogs	Not identified Not identified 1,000 Not identified	1,000 dog 1,000 monkey 5,000 rat 1,000 mouse

Table 4-36. NOAELs and LOAELs in animal studies involving inhalation exposure to dichloromethane for subchronic or chronic durations, hepatic, pulmonary, and neurologic effects

Type of effect and exposure period, reference	Species and exposure details	Results	NOAEL	LOAEL
			ppm	
Karlsson et al. (1987) Briving et al. (1986) Rosengren et al. (1986)	Mongolian gerbils, male and female 210, 350, 700 ppm (continuous exposure, followed by 4 month exposure-free period)	Astrogliosis in frontal and sensory motor cerebral cortex suggested by increases in astroglial proteins; cell loss in cerebellar regions; decreased DNA in hippocampus; neurochemical changes observed at all exposures	Not identified	210
Thomas et al. (1972)	ICR mice, female 0, 25, 100 ppm, continuous	Increased spontaneous activity observed at 25 ppm but not 100 ppm	Not identified	25
CoHb, 13-14 weeks				
Haun et al. (1972)	Beagles (n = 16); Rhesus monkeys (n = 4); Sprague-Dawley rats (n = 20), ICR mice (n = 20) 0, 25, 100 ppm (continuous exposure; 14 weeks)	CoHb levels significantly higher at 25, 100 ppm for monkeys and 100 ppm for beagles	Not identified	25
COHb, 2 years (6 hours/day, 5 day/week)				
Burek et al. (1984)	Syrian golden hamster, male and female 0, 500, 1,500, 3,500 ppm	About 30% COHb in each exposed group		
Burek et al. (1984)	Sprague-Dawley rat, male and female 0, 500, 1,500, 3,500 ppm	About 12–14% COHb in each exposed group		
Nitschke et al. (1988a)	Sprague-Dawley rat, male and female 0, 50, 200, 500 ppm	COHb values at 2 years: about 2, 7, 13, 14%		

[a]Equivalent to 1,000 ppm TWA.
[b]Decreased GSH, γ-aminobutyric acid, and phosphoethanolamine in frontal cortex; GSH and γ-aminobutyric acid increased in posterior cerebellar vermis.

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Table 4-37. NOAELs and LOAELs in selected animal studies involving inhalation exposure to dichloromethane, reproductive and developmental effects

Type of effect and exposure period, reference	Species and exposure details	Results	NOAEL ppm	LOAEL ppm
		Reproductive		
Nitschke et al. (1988b)	F344 rat, male and female, F0: 6 hr/d, 5 d/wk for 14 wk before mating and GDs 0 to 21; F1: 6 hr/d, 5 d/wk, beginning PND 4 for 17 wk before mating; 0, 100, 500, 1,500 ppm	No statistically significant effects on fertility or litter size, neonatal survival, growth rates, or histopathologic lesions in F1 or F2 weanlings	1,500	Not identified
Mennear et al. (1988); NTP (1986)	$B6C3F_1$ mouse; 0, 2,000 or 4,000 ppm, 6 hours/day, 5 days/week for 2 years	Testicular atrophy Ovarian atrophy (considered secondary to hepatic effects)	2,000 Not identified	4,000 2,000
Raje et al. (1988)	Swiss-Webster mouse, male, 2 hr/d, 5 d/wk for 6 wk before mating with nonexposed females; 0, 100, 150, 200 ppm	No statistically significant effects on testes, number of litters, live fetuses/litter, percent dead fetuses/litter, percent resorbed/litter	200	Not identified
		Fertility index was lower in 150 and 200 ppm groups (80%) compared with controls and 100 ppm groups (95%) (statistical significance depends on test used).	100	150
		Developmental		
Schwetz et al. (1975)	Swiss-Webster mouse, pregnant female, 7 hr/d, GDs 6–15; 0, 1,250 ppm	Maternal effects: 9–10% COHb; increased absolute, not relative, liver weight, increased maternal weight (11–15%).	Not identified	1,250
		Fetal effects: increased litters with extra center of ossification in sternum	1,250	Not identified
Schwetz et al. (1975)	Sprague-Dawley rat, pregnant female, 7 hr/d, GDs 6–15; 0, 1,250 ppm	Maternal: 9–10% COHb; increased absolute, not relative, liver weight	Not identified	1,250
		Fetal: increased incidence of delayed ossification of sternebrae	1,250	Not identified
		Other developmental		
Bornschein et al. (1980); Hardin and Manson (1980)	Long-Evans rat, female, 6 hr/d for 12–14 d before breeding and GDs 1–17; 6 hr/d on GDs 1–15; 0, 4,500 ppm	Maternal (both protocols): increased absolute and relative liver weight (~10%)	Not identified	4,500
		Fetal/offspring: decreased fetal BW (~10%); changed behavioral habituation to novel environments; no changes in gross, skeletal, or soft-tissue anomalies	Not identified	4,500

Hepatic centrilobular degeneration was observed in several studies containing different species and inhalational exposures. This effect was observed in guinea pigs exposed to 5,000 ppm (7 hours/day) for 6 months (Heppel et al., 1944). Monkeys, rats, and mice continuously exposed (24 hours/day) to 5,000 ppm dichloromethane for 14 weeks also had increased centrilobular degeneration (Haun et al., 1972, 1971). This effect was also observed at lower exposures when mice were exposed to 4,200 ppm for 6 hours/day for 13 weeks (NTP, 1986) and in dogs exposed to 1,000 ppm for 24 hours/day for up to 14 weeks (Haun et al., 1972, 1971).

Increased incidences of histologic hepatic lesions were not found in F344 rats exposed to 4,200 or 8,400 ppm 6 hours/day for 13 weeks (NTP, 1986) or in Sprague-Dawley rats exposed to 10,000 ppm 6 hours/day for 90 days (Leuschner et al., 1984). Hepatic lesions were also not observed in beagle dogs exposed to 5,000 ppm 6 hours/day for 90 days (Leuschner et al., 1984) or in dogs, monkeys, rats, and mice exposed to 25 or 100 ppm for 24 hours/day for up to 14 weeks (Haun et al., 1972). Heppel et al. (1944) also demonstrated absence of hepatic lesions in unspecified strains of monkeys, rabbits, and rats exposed to 10,000 ppm 4 hours/day for up to 8 weeks and in unspecified strains of dogs, rabbits, and rats exposed to 5,000 ppm 7 hours/day for up to 6 months.

Gross neurological impairments were observed in several laboratory species with repeated exposure to 10,000 ppm for 4 hours/day for 8 weeks (Heppel et al., 1944) or to 1,000 or 5,000 ppm for 24 hours/day for 14 weeks (Haun et al., 1972, 1971). Dogs exposed to 5,000 ppm 6 hours/day for 90 days showed slight sedation during exposures, but Sprague-Dawley rats exposed to 10,000 ppm for 90 days did not (Leuschner et al., 1984). In F344 rats exposed to concentrations up to 2,000 ppm 6 hours/day for 13 weeks, no effects were observed on an observational battery, hind-limb grip strength, a battery of evoked potentials, or histology of the brain, spinal cord, or peripheral nerves; these tests were conducted beginning 65 hours or more after the last exposure (Mattsson et al., 1990).

Exposure-related nonneoplastic effects on the lungs reported in the subchronic studies were restricted to foreign body pneumonia in rats exposed to 8,400 ppm 6 hours/day for 13 weeks (NTP, 1986), Clara cell vacuolation in mice exposed to 4,000 ppm 6 hours/day for 13 weeks (Foster et al., 1992), and pulmonary congestion in guinea pigs exposed to 5,000 ppm 7 hours/day for 6 months (Heppel et al., 1944).

The chronic duration inhalation studies were conducted at lower exposure levels than the short-term and subchronic studies and provide results indicating that the liver is the most sensitive target for noncancer toxicity in rats and mice (Table 4-36). Life-time exposure was associated with hepatocyte vacuolation and necrosis in F344 rats exposed to 1,000 ppm 6 hours/day (Mennear et al., 1988; NTP, 1986), hepatocyte vacuolation in Sprague-Dawley rats exposed to 500 ppm 6 hours/day (Nitschke et al., 1988a; Burek et al., 1984), and hepatocyte degeneration in $B6C3F_1$ mice exposed to 2,000 ppm 6 hours/day (lower concentrations were not

tested in mice) (Mennear et al., 1988; NTP, 1986). As shown in Tables 4-36 and 4-37, other effects observed include renal tubular degenerations in F344 rats and $B6C3F_1$ mice at 2,000 ppm, testicular atrophy in $B6C3F_1$ mice at 4,000 ppm, and ovarian atrophy in $B6C3F_1$ mice at 2,000 ppm (considered secondary to hepatic effects). No exposure-related increased incidences of nonneoplastic lung lesions were found in any of the chronic studies (Table 4-36).

In comparison to rats and mice, Syrian golden hamsters are less sensitive to the chronic inhalation toxicity of dichloromethane. No exposure-related changes were found in comprehensive sets of histologic, clinical chemistry, urinalytic, and hematologic variables measured in hamsters exposed for 2 years to 500, 1,500, or 3,500 ppm for 6 hours/day, with the exception that mean COHb percentages were about 30% in each of these groups compared with a mean value of about 3% for the controls (Burek et al., 1984).

The reproductive and developmental studies are limiting in terms of the exposure regimen used, with two of the developmental studies using only a single, relatively high daily exposure over the gestational period (1,250 ppm, GD 6-15 in Schwetz et al. [1975] and 4,500 ppm, GD 1-17 in Hardin and Manson [1980] and Bornschein [1980]). No significant effects on reproductive performance variables were found in a two-generation reproduction assay with F344 rats exposed to concentrations as high as 1,500 ppm (Nitschke et al., 1988b). No effects on most of the measures of reproductive performance were observed in male mice exposed to 200 ppm for 2 hours/day for 6 weeks before mating to nonexposed females. Fertility index was reduced in the 150 and 200 ppm groups, but the statistical significance of this effect varied considerably depending on the statistical test used in this analysis (Raje et al., 1988). No adverse effects on fetal development were found following exposure of pregnant Swiss-Webster mice or Sprague-Dawley rats to 1,250 ppm 6 hours/day on GDs 6–15 (Schwetz et al., 1975). Following exposure of female Long-Evans rats to 4,500 ppm (6 hours/day) for 14 days before breeding plus during gestation or during gestation alone, a 10% decrease in fetal BW and changed behavioral habituation of the offspring to novel environments were seen (Bornschein et al., 1980; Hardin and Manson, 1980). No exposure-related changes in gross, skeletal, or soft-tissue anomalies were found.

4.6.3. Mode of Action Information

4.6.3.1. *Mode of Action for Nonneoplastic Liver Effects*

Studies of chronically exposed rats, both by the oral route and the inhalation route, identified liver changes as the most sensitive exposure-related noncancer effect associated with exposure to dichloromethane (Tables 4-35 to 4-37). The liver changes included increased incidence of liver foci/areas of alteration and hepatocyte vacuolation in rats and degenerative liver effects in rats, guinea pigs, monkeys, and mice.

The mode of action by which dichloromethane induces these nonneoplastic hepatic effects is unknown. The determination of whether or not these effects are due to the parent

material, metabolites of the CYP2E1 pathway, metabolites of the GST pathway, or some combination of parent material and metabolites has not been elucidated. The available data indicate that rats may be more sensitive than mice to the noncancer hepatotoxicity, but a mechanistic explanation of this possible interspecies difference is not currently available.

4.6.3.2. *Mode of Action for Nonneoplastic Lung Effects*

Single 6-hour inhalation exposures to concentrations ≥2,000 ppm dichloromethane produced a transient vacuolation of Clara cells in the bronchiolar epithelium of $B6C3F_1$ mice. Vacuolization of the Clara cells disappeared or was diminished with repeated exposure and was correlated with subsequent transient diminishment of CYP metabolic activity. CYP inhibition with piperonyl butoxide counteracted the vacuolation observed in the Clara cells (Foster et al., 1994, 1992). With repeated exposure to 4,000 ppm (up to 13 weeks), the Clara cell vacuolation did not appear to progress to necrosis, and no hyperplasia of the bronchiolar epithelium was found. Foster et al. (1994, 1992) proposed that the diminished severity or disappearance of Clara cell vacuolation with repeated exposure was due to the development of tolerance to dichloromethane, linked with a transient decrease of CYP metabolism of dichloromethane. The available data suggests that CYP metabolism of dichloromethane may be involved in the mode of action for the acute effects of dichloromethane on the bronchiolar epithelium of mice.

Mode of action research attention on lung effects from chronic exposure to dichloromethane has focused on neoplastic effect; nonneoplastic lung effects have received relatively little attention. No exposure-related increased incidences of nonneoplastic lung lesions (including epithelial hyperplasia) were found in any of the chronic studies listed in Table 4-36, but chronic inhalation exposure of $B6C3F_1$ mice to concentrations ≥2,000 ppm has consistently been shown to induce lung tumors in several studies (Kari et al., 1993; NTP, 1986). In a study that included interim sacrifices at 13, 26, 52, 68, 75, 78, 83, and 91 weeks of $B6C3F_1$ mice exposed to 2,000 ppm, hyperplasia of lung epithelium (the only nonneoplastic lung lesion found) was found in only three of the eight interim sacrifices (68, 78, and 91 weeks) and was only statistically significantly elevated at 91 weeks (5/30 versus 0/15 in controls) (Kari et al., 1993).

4.6.3.3. *Mode of Action for Neurological Effects*

Results from studies of acutely exposed human subjects indicate that mild neurobehavioral deficits may occur at air concentrations >200 ppm with 4–8 hours of exposure (Bos et al., 2006; ACGIH, 2001; ATSDR, 2000; Cherry et al., 1983; Putz et al., 1979; Gamberale et al., 1975; Winneke, 1974). Acute high-dose exposures also resulted in gross neurological impairments in several laboratory species (Haun et al., 1972, 1971; Heppel et al., 1944). Exposure of F344 rats to concentrations up to 2,000 ppm 6 hours/day for 13 weeks produced no effects on an observational battery, hind-limb grip strength, a battery of evoked potentials, or histology of the brain, spinal cord, or peripheral nerves (Mattsson et al., 1990). However, oral

exposures have been shown to alter autonomic, neuromuscular, and sensorimotor functions have been observed in F344 rats exposed to gavage doses ≥337 mg/kg-day for 14 days (Moser et al., 1995).

Dichloromethane may be metabolized by the CYP2E1 enzyme to CO (Guengerich, 1997; Hashmi et al., 1994; Gargas et al., 1986). Many of the acute human exposure studies evaluated if CO was the primary metabolite responsible for producing the CNS depressant effects observed during dichloromethane exposure. Overall, at lower exposures and acute durations, it appears that CO is the primary mediator of the neurobehavioral effects. Putz et al. (1979) demonstrated that similar neurobehavioral deficits were present when an equivalent COHb blood level (and CO exposure) was achieved between CO and dichloromethane exposures. Incidentally, after a longer duration, neurobehavioral deficits are more pronounced with dichloromethane exposure in comparison to CO exposure alone. This additional increase in the CNS depressive effects is most likely due to the saturation of the CYP2E1 metabolic pathway. In humans, saturation of the CYP2E1 metabolic pathway was seen at approximately 400–500 ppm after a 1-hour exposure (Ott et al., 1983e). CYP2E1 pathway saturation with dichloromethane has also been noted in hamsters (Burek et al., 1984) and in rats (McKenna et al, 1982; Nitschke et al., 1988a). It is highly probable that initially CYP2E1 is metabolizing dichloromethane to CO, which results in the neurological effects. However, at higher concentrations (greater than 500 ppm) and for longer durations (greater than 3 hours), the CYP2E1 pathway is most likely saturated. As a result, either the remaining dichloromethane could be metabolized by the GST pathway or the parent compound is producing the effects itself.

Once the CYP2E1 enzyme is saturated, it is unknown whether dichloromethane or a GST-T1 pathway metabolite (e.g., formaldehyde) mediates the resulting neurological effects. Based on the available literature on other solvents, such as toluene and perchloroethylene (for a review see Bowen et al. [2006]), it can be hypothesized that once the CYP2E1 enzyme is saturated dichloromethane or a GST metabolite may interact directly with excitatory and inhibitory receptors, such as the NMDA, GABA, dopamine, and serotonin receptors among other targets, to produce the resulting neurobehavioral effects. This hypothesis is supported by the evidence that changes in relation to dichloromethane exposures in glutamate, GABA, dopamine, serotonin, acetylcholine, and other neurotransmitters are found in the brain (Kanada et al., 1994; Briving et al., 1986; Fuxe et al., 1984). Additionally, several neurobehavioral effects, such as decreased spontaneous motor activity, deficits in learning and memory, and deficits in FOB parameters are similar to other more characterized solvents such as toluene. However, more comprehensive studies specifically designed to determine the mode of action for dichloromethane-induced impairment of neurological functions have not been conducted.

4.6.3.4. *Mode of Action for Reproductive and Developmental Effects*

No significant effects on reproductive performance variables were found in a two-generation reproduction assay with F344 rats exposed to concentrations as high as 1,500 ppm (Nitschke et al., 1988b), and no effects were seen on most of the measures of reproductive performance examined in a study of male mice exposed to 200 ppm for 2 hours/day for 6 weeks before mating to nonexposed females (Raje et al., 1988). In the mouse study, fertility index (number of females impregnated divided by total number of females mated times 100) was reduced in the 150 and 200 ppm groups (Raje et al., 1988), but the statistical significance of this effect varied considerably depending on the statistical test used in the analysis. Mechanistic studies of dichloromethane or its metabolites that would provide mode of action information on reproductive effects in the male are not available.

The mode of action for developmental effects can be hypothesized to involve the CYP2E1 pathway and, specifically, the production of CO. CO is a known developmental neurotoxicant. Demonstrated effects include neurobehavioral deficits and neurochemical changes (Giustino et al., 1999; Cagiano et al., 1998; De Salvia et al., 1995; Fechter, 1987). In addition, placental transfer of dichloromethane has been demonstrated with inhalation exposure (Withey and Karpinski, 1985; Anders and Sunram, 1982). Pups exposed in utero to high concentrations of dichloromethane (4,500 ppm) demonstrated neurobehavioral-related changes in comparison to air-exposed animals (Bornschein et al., 1980). This observed effect coupled with the known developmental neurotoxicological effects produced by CO suggests that the CYP2E1 metabolic pathway is involved in producing observed and suspected neurodevelopmental effects. In humans, CYP2E1 activity in the brain occurs earlier in gestation than it does in the liver, with activity in the brain seen in the first trimester (Johnsrud et al., 2003; Brzezinski et al., 1999). Thus, the direct effects of dichloromethane in fetal circulation, as well as the effects of CO and the effects of the CYP2E1-related metabolism in the fetal liver and the fetal brain, may be relevant to the risk of developmental effects in humans. Mechanistic studies of dichloromethane or its metabolites that would provide mode of action information on other noted developmental effects such as delayed ossification (Schwetz et al., 1975) are not available.

4.6.3.5. *Mode of Action for Immunotoxicity*

Evidence of a localized immunosuppressive effect in the lung resulting from inhalation dichloromethane exposure was seen in an acute exposure (3 hours, 100 ppm) study in CD-1 mice (Aranyi et al., 1986). The lung infectivity assay used in this study examined response to bacterial challenges (i.e., risk of streptococcal-pneumonia-related mortality and clearance of Klebsiella bacteria). The innate immune response plays an important role in limiting the initial lung burden of bacteria through the activity of macrophages, neutrophils, and dendritic cells**,** and alveolar macrophages are particularly important in the response to respiratory infections (Marriott and Dockrell, 2007). The adaptive response develops from several days up to several weeks following infection, so that an effective immune response in a lung infectivity assay

requires multiple immune mechanisms and in particular cooperation of macrophages, neutrophils, and T cells along with the appropriate cytokines (Selgrade and Gilmour, 2006). Although immunosuppression in the Streptococcal and Klebsiella infectivity models has been reported in the acute exposure scenarios tested in Aranyi et al. (1986), mechanistic studies of dichloromethane or its metabolites that would provide mode of action information on the immune system cells or function have not been performed.

4.7. EVALUATION OF CARCINOGENICITY

4.7.1. Summary of Overall Weight of Evidence

Following U.S. EPA (2005a) *Guidelines for Carcinogen Risk Assessment*, dichloromethane is "likely to be carcinogenic in humans" by the inhalation and oral routes of exposure, based predominantly on evidence of carcinogenicity at two sites in 2-year bioassays in B6C3F_1 mice (liver and lung tumors with inhalation exposure in both sexes, liver tumors with drinking water exposure in males only). In addition, evidence of a trend for increased risk of liver tumors (described as neoplastic nodule or hepatocellular carcinoma) was seen in female F344 rats exposed via drinking water ($p < 0.01$) (Serota et al., 1986a) or inhalation ($p = 0.08$) (NTP, 1986). However, the potential malignant characterization of the nodules was not described, and no trend was seen in the data limited to hepatocellular carcinomas. Additional evidence of the tumorigenic potential of dichloromethane comes from the observation of an increase in benign mammary tumors following inhalation exposure (NTP, 1986; Burek et al., 1986b; Nitschke et al. 1988a). An inhalation study (exposures of 0, 50, 200, and 500 ppm) also reported the presence of another relatively rare tumor in rats, astrocytoma or glioma (mixed glial cell) tumors (Nitschke et al., 1988a). This collection of studies in the rat does not provide evidence for a carcinogenic response that is as strong as that seen in the mouse. Taken together, however, the rat data provide supporting evidence of carcinogenicity. Studies in humans found some evidence linking occupational exposure to dichloromethane and increased risk for some specific cancers, including brain cancer (Hearne and Pifer, 1999; Tomenson et al., 1997; Heineman et al., 1994) and liver cancer (Lanes et al., 1993, 1990).

The proposed mode of action for dichloromethane-induced liver tumors is through a mutagenic mode of carcinogenic action. Mode of action data indicate that dichloromethane-induced DNA damage in cancer target tissues of mice involves DNA-reactive metabolites produced via a metabolic pathway initially catalyzed by GST-T1. Evidence of mutagenicity includes in vitro bacterial and mammalian assays as well as in vivo mammalian system assays, although mutational events in critical genes (tumor suppressor genes, oncogenes) leading to tumor initiation and tumor promotion have not been established. This metabolic pathway has been found in human tissues, albeit at lower activities than in mouse tissues; therefore, the cancer results in animals are considered relevant to humans.

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

Section 4.1.2 reviewed the results, strengths, and limitations of epidemiological research of dichloromethane and cancer, including cohort and case-control studies. The available epidemiologic studies provide some evidence of an association between dichloromethane and brain cancer and liver cancer, but the available data are limited.

Two small cohort studies with relatively good exposure metrics and relatively long follow-up periods (mean over 25 years) reported an increased risk of brain cancer, with SMRs of 1.45 (95% CI 0.40–3.72) in Tomenson et al. (1997) and 2.2 (95% CI 0.79–4.69) in Cohort 1 of Hearne and Pifer (1999). Cohort 1 is an inception cohort, following workers from the beginning of their employment, which is methodologically more robust than Cohort 2, which only included workers who were working between 1964 and 1970. These observations are supported by the data from a case-control study of brain cancer that reported relatively strong trends with increasing probability, duration, and intensity measures of exposure but not with a cumulative exposure measure. This difference could reflect a relatively more valid measure of relevant exposures in the brain from the intensity measure, as suggested by the study in rats reported by Savolainen et al. (1981) in which dichloromethane levels in the brain were much higher with a higher intensity exposure scenario compared with a constant exposure period with an equivalent TWA (see section 3.2). A statistically significant increased incidence of brain or CNS tumors has not been observed in any of the animal cancer bioassays, but a 2-year study using relatively low exposure levels (0, 50, 200, and 500 ppm) in Sprague-Dawley rats observed a total of six astrocytoma or glioma (mixed glial cell) tumors in the exposed groups (in females, the incidence was 0, 0, 0, and 2 in the 0, 50, 200, and 500 ppm exposure groups; in males, the incidence was 0, 1, 2, and 1 in the 0, 50, 200, and 500 ppm exposure groups; sample size of each group was 70 rats). These tumors are exceedingly rare in rats, and there are few examples of statistically significant trends in animal bioassays (Sills et al., 1999). These cancers were not seen in two other studies in rats, both involving higher doses (1,000–4,000 ppm) (NTP, 1986; Burek et al., 1984), or in a high dose (2,000–4,000 ppm) study in mice (NTP, 1986).

With respect to epidemiologic studies of liver and biliary duct cancer, the highest exposure cohort, based in the Rock Hill, South Carolina, triacetate fiber production plant, suggested an increased risk of liver cancer, with an SMR of 2.98 (95% CI 0.81, 7.63) in the latest study update (Lanes et al., 1993). This observation was based on four cases; an earlier analysis in this cohort reported an SMR of 5.75 (95% CI 1.82, 13.8), based on these same four cases but with a shorter follow-up period (and thus a lower number of expected cases) (Lanes et al., 1990). No other cohort study has reported an increased risk of liver cancer mortality, although it should be noted that there is no other inception cohort study of a population with exposure levels similar to those of the Rock Hill plant, and no data from a case-control study of liver cancer are available pertaining to dichloromethane exposure.

The primary limitation of all of the available dichloromethane cohort studies is the limited statistical power for the estimation of effects relating to relatively rare cancers (such as brain cancer, liver cancer, and leukemia). Limitations with respect to studies of other cancers can also be noted. With respect to breast cancer, the only cohort that included a significant percentage of women had limited exposure information (analysis was based on a dichotomous exposure variable) and co-exposure to other solvents (Blair et al., 1998). The only breast cancer case-control study available used death certificate data to classify disease and occupational exposure (Cantor et al., 1995), which is likely to result in significant misclassification; exposure misclassification in particular would be expected to result in an attenuated measure of association (Rothman and Greenland, 1998). No studies of adult leukemia and dichloromethane exposure and only one study of childhood leukemia (acute lymphoblastic leukemia) in relation to maternal occupational dichloromethane exposure were found. Thus, EPA views the epidemiologic data pertaining to breast cancer and leukemia as inadequate to assess carcinogenic potential.

In addition to the epidemiologic studies, several dichloromethane cancer bioassays in animals are available. In the only oral exposure cancer bioassay involving lifetime exposure, increases in incidence of liver adenomas and carcinomas were observed in male (trend *p*-value = 0.058) but not female $B6C3F_1$ mice exposed for 2 years (Table 4-38) (Serota et al., 1986b; Hazelton Laboratories, 1983). The authors concluded that these increases were "within the normal fluctuation of this type of tumor incidence," noting that there was no dose-related trend and that most of the individual group paired tests were not statistically significant after use of a Bonferroni correction factor. [The trend *p*-value and pairwise test *p*-values were not given in the Serota et al. (1986b) paper but can be found in the full report (Hazelton Laboratories, 1983)]. However, the trend *p*-value for these results is of borderline statistical significance and it may not be reasonable to apply a correction for multiple comparisons given the lack of independence of the groups and given a specific focus on the liver as a target organ. In Syrian golden hamsters exposed to 500, 1,500, or 3,500 ppm for 2 years, no statistically significantly increased incidences of tumors were found in any tissues (Burek et al., 1984).

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Table 4-38. Incidence of liver tumors in male B6C3F$_1$ mice exposed to dichloromethane in a 2-year oral exposure (drinking water) study[a]

Estimated mean intake (mg/kg-day)[a]	Controls 0	61	124	177	234	Trend *p*-value[d]
Number of male mice[b]	125[c]	200	100	99	125	
			Number of cancers (%)			
Hepatocellular adenoma or carcinoma	24 (19)	51 (26)	30 (30)	31 (31)	35 (28)	0.058
Mortality-adjusted percent[e]	(22)	(29)	(34)	(35)	(32)	
Mortality-adjusted *p*-value[e]		$P = 0.071$	$p = 0.023$	$p = 0.019$	$p = 0.036$	

[a]Target doses were 60, 125, 185, and 250 mg/kg-day from the lowest dose group (excluding controls) to the highest dose group, respectively.
[b]No significant increases in females were found, but incidence data were not reported.
[c]Two control groups combined. The incidence in control groups 1 and 2 were 20 and 23%, respectively.
[d]Cochran-Armitage trend test (source: Hazelton Laboratories [1983]).
[e]Mortality-adjusted percent calculated, based on number at risk, using Kaplan-Meier estimation, taking into account mortality losses; *p*-value for comparison with control group, using asymptotic normal test (source: Hazelton Laboratories [1983]).

Sources: Serota et al. (1986b); Hazelton Laboratories (1983).

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6054
6055 In a similar study in F344 rats (Serota et al., 1986a), no increased incidence of liver
6056 tumors was seen in male rats, and the pattern in female rats was characterized by a jagged
6057 stepped pattern of increasing incidence of hepatocellular carcinoma or neoplastic nodule
6058 (Table 4-39). Information was not provided which would allow characterization of the nodules
6059 as benign or malignant. Statistically significant increases in incidences were observed in the 50
6060 and 250 mg/kg-day groups (incidence rates of 0, 3, 10, 3, and 14%, respectively, for the 0, 5, 50,
6061 125, and 250 mg/kg-day groups) and in the group exposed to 250 mg/kg-day for 78 weeks
6062 followed by a 26-week period of no exposure (incidence rate 10%). A similar pattern, but with
6063 more sparse data, was seen for hepatocellular carcinomas, with 2 incidences in the 50 mg/kg-day
6064 and 2 in the 250 mg/kg-day groups. The authors concluded that dichloromethane exposure did
6065 not result in an increased incidence of liver tumors, because the increase was based on a low rate
6066 (0%) in the controls and because of a lack of monotonicity.

6067

Table 4-39. Incidences of liver tumors in male and female F344 rats exposed to dichloromethane in drinking water for 2 years

	Target dose (mg/kg-day)						
	Controls 0[a]	5	50	125	250	Trend *p*-value[b]	250 with recovery[c]
Males							
Estimated mean intake (mg/kg-day)	0	6	52	125	235		232
n per group[d]	76	34	38	35	41		15
Number (%) with neoplastic nodules	9 (12)	1 (3)	0 (0)	2 (6)	1 (2)	Not reported	2 (13)
Number (%) with hepatocellular carcinoma	3 (4)	0 (0)	0 (0)	0 (0)	1 (2)	Not reported	0 (0)
Number (%) with neoplastic nodules and hepatocellular carcinoma	12 (16)	1 (3)	0 (0)	2 (6)	2 (5)	Not reported	2 (13)
Females							
Estimated mean intake (mg/kg-day)	0	6	58	136	263		239
n per group[d]	67	29	41	38	34		20
Number (%) with neoplastic nodules	0 (0)	1 (3)	2 (5)	1 (3)	3 (9)		2 (10)[e]
Number (%) with hepatocellular carcinoma	0 (0)	0 (0)	2 (5)	0 (0)	2 (6)	Not reported	0 (0)
Number (%) with neoplastic nodules and hepatocellular carcinoma	0 (0)	1 (3)	4 (10)[e]	1 (3)	5 (14)[e]	$p < 0.01$	2 (10)[e]

[a]Two control groups combined.
[b]Cochran-Armitage trend test was used for trend test of liver foci/areas of alteration. For tumor mortality-unadjusted analyses, a Cochran-Armitage trend test was used, and, for tumor mortality-adjusted analyses, a tumor prevalence analytic method by Dinse and Lagakos (1982) was used. Similar results were seen in these two analyses.
[c]Recovery group was exposed for 78 weeks and then had a 26-week period without dichloromethane exposure; n = 17 for neoplastic lesions.
[d]n available at terminal sacrifice; starting with 135 controls (combining both control groups) and 85 per sex per dose group except recovery group (n = 25); subtracted 5, 10, and 20 per group (except for recovery group) sacrificed at 25, 52, and 78 weeks, respectively, and subtracted unscheduled deaths, which ranged from 5 to 19 per group.
[e]Significantly ($p < 0.05$) different from controls with Fisher's exact test, mortality-unadjusted and mortality-adjusted analyses.

Source: Serota et al. (1986a).

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6070 Another oral (gavage) exposure study in Sprague-Dawley rats and in Swiss mice provides
6071 limited data concerning cancer incidence because the study was terminated early (at 64 weeks)
6072 due to high treatment-related mortality (Maltoni et al., 1988). Exposure groups included controls
6073 (olive oil), 100, or 500 mg/kg-day 4–5 days/week. High-dose female rats showed an increased
6074 incidence of malignant mammary tumors, mainly adenocarcinomas (8, 6, and 18% in the control,
6075 100, and 500 mg/kg dose groups, respectively), but the increase was not statistically significant.
6076 Data were not provided to allow an analysis accounting for differing mortality rates. A dose-
6077 related increase, although not statistically significant, in pulmonary adenomas was observed in

male mice (5, 12, and 18% in control, 100, and 500 mg/kg-day groups, respectively). When mortality was taken into account, high-dose male mice that died in the period ranging from 52 to 78 weeks were reported to show a statistically significantly ($p < 0.05$) elevated incidence for pulmonary tumors (1/14, 4/21, and 7/24 in control, 100, and 500 mg/kg-day groups, respectively). Details of this analysis were not provided. EPA applied a Fisher's exact test to these incidences and determined a p-*value* of 0.11 for the comparison of the 500 mg/kg-day group (7/24) to the controls (1/14).

As discussed in section 4.2, repeated inhalation exposure to concentrations of 2,000 or 4,000 ppm dichloromethane produced increased incidences of lung and liver tumors in $B6C3F_1$ mice (Mennear et al., 1988; NTP, 1986). The incidence of mortality-adjusted liver tumors across dose groups (0, 2,000, and 4,000 ppm) increased from 48 to 67 and 93%, respectively, in male mice (trend *p*-value = 0.013) and from 10 to 48 and 100% female mice (trend *p*-values <0.001) (Table 4-40). For lung tumors, the mortality-adjusted incidence was 12, 74, and 100% in males and 11, 83, and 100% in females in the 0, 2,000, and 4,000 ppm groups, respectively (trend *p*-values <0.001). Elevated incidences of lung and liver tumors in $B6C3F_1$ mice were observed with 52 weeks of exposure to 2,000 ppm, and lung tumors were also elevated by week 104 in mice exposed for only 26 weeks to 2,000 ppm, followed by 78 weeks without exposure (Maronpot et al., 1995; Kari et al., 1993).

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Table 4-40. Incidences of selected neoplastic lesions in B6C3F$_1$ mice exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

	Exposure (ppm)[a]									
	0 (Controls)			2,000			4,000			
Sex and neoplastic lesion	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	Trend *p*-value[d]
Males										
Liver—hepatocellular adenoma or carcinoma	22	(44)	(48)	24	(49)	(67)	33[e]	(67)	(93)	0.013
Lung—bronchoalveolar adenoma or carcinoma	5	(10)	(12)	27[e]	(54)	(74)	40[e]	(80)	(100)	<0.001
Females										
Liver— hepatocellular adenoma or carcinoma	3	(6)	(10)	16[e]	(33)	(48)	40[e]	(83)	(100)	<0.001
Lung—bronchoalveolar adenoma or carcinoma	3	(6)	(11)	30[e]	(63)	(83)	41[e]	(85)	(100)	<0.001

[a]2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.

[b]Total sample size was 50 per sex and dose group. Percentages based on the number of tissues examined microscopically per group; for male mice, 49 livers were examined in the 2,000 and 4,000 ppm groups; for female mice, 48 liver and lungs were examined. For comparison, incidence in historical controls reported in NTP (1986) were 28% for male liver tumors, 31% for male lung tumors, 5% for female liver tumors, and 10% for female lung tumors.

[c]Mortality-adjusted percentage.

[d]Life-table trend test, as reported by NTP (1986).

[e]Life-table test comparison dose group with control <0.05, as reported by NTP (1986).

Sources: Mennear et al. (1988); NTP (1986).

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6100
6101 Liver tumors are relatively rare in F344 rats, and a moderate trend of increasing incidence
6102 of what was described as neoplastic nodules or hepatocellular carcinoma was seen in the females
6103 (trend *p*-value = 0.08) but not the males in the NTP (1986) study (Table 4-41). As with the rat
6104 oral exposure study by Serota et al., 91986a), these nodules were not characterized as benign or
6105 malignant. There was no evidence of an increasing trend in incidence when hepatocellular
6106 carcinomas only were considered.
6107 Female F344 rats exposed by inhalation to 2,000 or 4,000 ppm showed significantly
6108 increased incidences of benign mammary tumors (adenomas or fibroadenomas) (Table 4-41); the
6109 number of benign mammary tumors per animal also increased with dichloromethane exposure in
6110 studies in Sprague-Dawley rats at levels of 50–500 ppm (Nitschke et al., 1988a) and 500–3,500
6111 ppm (Burek et al., 1984) (Table 4-42). Male rats in two of these studies (Nitscke et al., 1988a ;
6112 NTP, 1986) also exhibited a low rate of sarcoma or fibrosarcoma in mammary gland or
6113 subcutaneous tissue around the mammary gland.
6114

Table 4-41. Incidences of selected neoplastic lesions in F344/N rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

Sex and neoplastic lesion	Exposure (ppm)[a]												
	0 (Controls)			1,000			2,000			4,000			
	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	n	(%)[b]	(%)[c]	Trend *p*-value[d]
Males													
Liver—Neoplastic nodule or hepatocellular carcinoma	2	(4)	(10)	3	(6)	(13)	4	(8)	(19)	1	(2)	(6)	0.55
Liver—hepatocellular carcinoma	2	(4)	(10)	1	(2)	(4)	2	(4)	(10)	1	(2)	(6)	nr
Lung—bronchoalveolar adenoma or carcinoma	1			1	(2)		2	(4)		1	(2)		
Mammary gland													
Adenoma, adenocarcinoma, or carcinoma	0	(0)		0	(0)		0	(0)		1	(2)		
Subcutaneous tissue fibroma or sarcoma	1	(2)	(6)	1	(2)	(6)	2	(4)	(9)	5	(10)	(23)	0.008
Fibroadenoma	0	(0)	(0)	0	(0)	(0)	2	(4)	(12)	1	(2)	(8)	< 0.001
Mammary gland or subcutaneous tissue adenoma, fibroadenoma, fibroma, or sarcoma	1	(2)	(6)	1	(2)	(6)	4	(8)	(21)	9[d]	(18)	(49)	<0.001
Females													
Liver—neoplastic nodule or hepatocellular carcinoma	2	(4)	(7)	1	(2)	(2)	4	(8)	(14)	5	(10)	(20)	0.08
Liver—hepatocellular carcinoma	0	(0)	(0)	0	(0)	(0)	1	(2)	(4)	0	(0)	(0)	nr
Lung—bronchoalveolar adenoma or carcinoma	1	(2)		1	(2)		0	(0)		0	(0)		
Mammary gland													
Adenocarcinoma or carcinoma	1	(2)		2	(4)		2	(4)		0	(0)		
Adenoma, adenocarcinoma, or carcinoma	1	(2)		2	(4)		2	(4)		1	(2)		
Fibroadenoma	5	(10)	(16)	11[d]	(22)	(41)	13[d]	(26)	(44)	22[d]	(44)	(79)	<0.001
Mammary gland adenoma, fibroadenoma, or adenocarcinoma	6	(12)	(18)	13	(26)	(44)	14[d]	(28)	(45)	23[e]	(46)	(86)	<0.001

[a]1,000 ppm = 3,474 mg/m^3, 2,000 ppm = 6,947 mg/m^3, 4,000 ppm = 13,894 mg/m^3.

[b]Total sample size was 50 per sex and dose group. Percentages based on the number of tissues examined microscopically per group; for male rats, 49 livers were examined in the 2,000 and 4,000 ppm groups; for females, only 48 liver and lungs and 49 mammary glands were microscopically examined in the 2,000 and 4,000 ppm groups. For comparison, incidence in historical controls reported in NTP (1986) were 1% for female liver tumors and 16% for female mammary fibroadenomas.

[c]Mortality-adjusted percentage.

[d]Life-table trend test, as reported by NTP (1986). nr = not reported.

[e]Life-table test comparison dose group with control <0.05, as reported by NTP (1986).

Sources: Mennear et al. (1988); NTP (1986).

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Table 4-42. Incidences of mammary gland tumors in two studies of male and female Sprague-Dawley rats exposed to dichloromethane by inhalation (6 hours/day, 5 days/week) for 2 years

Study, lesion	Exposure (ppm)[a] Controls 0	50	200	500	Late 500[b]	Early 500[b]	1,500	3,500
Nitschke et al. (1988a)								
Males—n per group	57	65	59	64	[c]	[c]	[c]	[c]
Number (%) with								
Mammary gland tumors								
Adenocarcinoma or carcinoma	0 (0)	0 (0)	0 (0)	0 (0)				
Fibroadenoma	2 (4)	0 (0)	2 (3)	2 (3)				
Fibroma	6 (11)	1 (6)	6 (11)	10 (16)				
Fibrosarcoma	0 (0)	1 (6)	1 (6)	0 (0)				
Undifferentiated sarcoma	0 (0)	2 (4)	0 (0)	0 (0)				
Fibroma, fibrosarcoma, or undifferentiated sarcoma[d]	6 (11)	4 (6)	7 (12)	10 (16)				
Females—n per group	69	69	69	69	25	25	[c]	[c]
Number (%) with mammary gland								
Mammary gland tumors								
Adenocarcinoma or carcinoma	6 (9)	5 (7)	4 (6)	4 (6)	3 (12)	2 (8)		
Adenoma	1 (1)	1 (1)	2 (3)	1 (1)	2 (8)	0 (0)		
Fibroadenoma	51 (74)	57 (83)	60 (87)	55 (80)	22 (88)	23 (92)		
Fibroma	0 (0)	1 (1)	0 (0)	1 (1)	1 (4)	1 (1)		
Fibrosarcoma	1 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)		
Number with benign tumors[e]	52 (74)	58 (83)	61(87)[f]	55 (79)	23 (92)	23 (92)		
Number of benign tumors per tumor-bearing rat[e]	2.0	2.3	2.2	2.7	2.2	2.6		
Burek et al. (1984)								
Males								
n per group	92	[c]	[c]	95	[c]	[c]	96	97
Number (%) with benign tumors	7 (8)			3 (3)			7 (7)	14 (14)
Total number of benign tumors	8			6			11	17
Number of tumors per tumor-bearing rat[g]	1.1			2.0			1.6	1.2
Females								
n per group	96	[c]	[c]	95	[c]	[c]	96	97
Number (%) with benign tumors	79 (82)			81 (85)			80 (83)	83 (86)
Total number of benign tumors	165			218			245	287
Number of tumors per tumor-bearing rat[f]	2.1			2.7			3.1	3.5

[a]50 ppm = 174 mg/m^3, 200 ppm = 695 mg/m^3, 500 ppm = 1,737 mg/m^3, 1,500 ppm = 5,210 mg/m^3, 3,500 ppm = 12,158 mg/m^3.
[b]Late 500 = no exposure for first 12 months followed by 500 ppm for last 12 months; early 500 = 500 ppm for first 12 months followed by no exposure for last 12 months.
[c]No data for this exposure level in this study.
[d]EPA summed across these tumor types, assuming no overlap.
[e]In historical controls, percent with benign tumors reported as 79–82% and number per tumor-bearing rat was 2.1.
[f]Significantly ($p \leq 0.05$) higher than control incidence by Fisher's exact test (Nitschke et al., 1988a).
[g]Calculated by EPA.

Sources: Nitschke et al. (1988a); Burek et al. (1984b).

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Supporting evidence for the carcinogenicity of dichloromethane comes from the results of genotoxicity and mode of action studies discussed in section 4.5. A mutagenic mode of carcinogenic action for dichloromethane involves metabolic activation by GST, as evidenced by several observations, including the enhancement of dichloromethane mutagenic activity in normally unresponsive *S. typhimurium* strain TA1535 after it is transfected with the gene for rat GST-T1 (DeMarini et al., 1997; Thier et al., 1993); increased HPRT gene mutations and DNA damage (DNA SSBs) in CHO cells when they are incubated with dichloromethane in the presence of mouse liver cytosol preparations rich in GST enzymatic activities (Graves and Green, 1996; Graves et al., 1996, 1994b); the detection of DNA damage (DNA SSBs) in liver and lung tissue of $B6C3F_1$ mice immediately following 6-hour inhalation exposure to dichloromethane (2,000–8,000 ppm); and a suppression of the DNA damage when mice were pretreated with buthionine sulphoximine, a GSH depletor (Graves et al., 1995).

Additional data from several studies indicate that dichloromethane genotoxicity is expressed in cancer target tissues in mice following in vivo exposure. Increased sister chromatid exchanges were observed in lung cells of $B6C3F_1$ mice after 90 days of inhalation exposure to 2,000 ppm or 10 days of exposure to 4,000 or 8,000 ppm (Allen et al., 1990). DNA damage (comet assay) was detected in liver and lung tissue (but not stomach, kidney, brain, or bone marrow) 24 hours after oral administration of 1,720 mg/kg dichloromethane to CD-1 mice (Sasaki et al., 1998). DNA-protein cross-links were observed in the liver of $B6C3F_1$ mice but not hamsters, following inhalation exposure to concentrations ranging from 500 to 4,000 ppm 6 hours/day for 3 days (Casanova et al., 1996, 1992). Much less is known about genotoxicity in the liver in rats. Studies of single-strand DNA breaks in rat hepatocytes or liver homogenate were negative, with inhalation exposures up to 5,000 ppm for 3 hours (Graves et al., 1995, 1994b), but positive results were seen in a high-dose gavage study (1,275 mg/kg) (Kitchin and Brown, 1989). Few other specific types of genotoxicity endpoints (e.g., sister chromatid exchange, DNA-protein cross-links) have been studied in the rat liver.

Since there are limited data on mutagenic events following oral exposure, EPA conducted a pharmacokinetic analysis to evaluate how comparable the internal doses to the liver in the oral bioassay (Serota et al., 1986b; Hazelton Laboratories, 1983) were to the internal doses to the liver in the inhalation bioassay (Mennear et al., 1988; NTP, 1986). The PBTK model of Marino et al. (2006) predicted that the average daily amount of dichloromethane metabolized via GST per liter of liver was about 14-fold lower in mice exposed to the highest dose of 244 mg/kg-day in the drinking water bioassay than in mice exposed to the lowest inhalation exposure of 2,000 ppm, inducing liver tumors (Table 4-43). Thus, the lower incidence of liver tumors induced by oral doses of 244 mg/kg-day, compared with the higher incidence induced by inhalation exposure to 2,000 ppm, is consistent with the predicted lower liver dose of GST metabolites (and hence lower probability of DNA modification) with oral exposure.

Table 4-43. Comparison of internal dose metrics in inhalation and oral exposure scenarios, in male mice and rats

	Internal exposure in liver (mg metabolized through GST pathway/L liver tissue/day)[a]	
	Male	
External dose	**Mouse**	**Rat**
Inhalation (ppm)		
2,000	2,364	1,502
4,000	4,972	3,111
Oral (mg/kg-day)[b]		
61	17.5	77.0
124	63.3	233.5
174	112.0	385.4
234	169.5	589.8

[a] Mouse values derived by EPA from the PBTK model of Marino et al. (2006); rat values derived from EPA based on the modified PBTK model of Andersen et al. (1991) (see Appendix C for model details).
[b] Actual doses administered to mice (Serota et al., 1986a); BWs not given for males and females, so simulation results only provided for one gender.

4.7.3. Mode of Action Information

4.7.3.1. *Hypothesized Mode of Action*

The hypothesized mode of action for dichloromethane-induced tumors is through a mutagenic mode of carcinogenic action. Specifically, the data indicate that dichloromethane is metabolized by GST to reactive metabolites that induce mutations in DNA leading to carcinogenicity. Much of the experimental mode of action research has focused on the liver and lung, the sites of tumor formation in chronic bioassays (Mennear et al., 1988; NTP, 1986; Serota et al., 1986b, Hazleton Laboratories, 1983). The mode of action is potentially relevant to other sites, particularly those in which GST-T1 is expressed, such as mammary tissue (Lehmann and Wagner, 2008) and the brain (Juronen et al., 1996).

Support for the importance of GST in the hypothesized mutagenic mode of action has been demonstrated in in vitro bacterial and mammalian assays as well as in vivo mammalian system assays. Dichloromethane is consistently mutagenic in *S. typhimurium* strains with GST capability, but did not produce mutagenic effects in non-GST *S. typhimurium* strains (summarized in Section 4.5.1.1 and Table 4-29). In vitro mammalian cell studies (see Table 4-30) have consistently demonstrated genotoxic effects in CHO cell lines when a mouse liver cytosol fraction was exogenously added and in mouse Clara cells; positive responses were seen in studies measuring DNA-protein crosslinks, HPRT mutation analysis, and DNA SSBs. Other

studies have demonstrated DNA adducts with dichloromethane exposure in calf thymus DNA in the presence of bacterial GST DM11. Negative results were seen in most of the other in vitro cell studies using rat hepatocytes or CHO cells without mouse liver cytosol incubation. These studies were conducted in cell lines where GST activity is considerably lower than in mouse cell lines and therefore these results are not unexpected.

In studies with human cell lines or isolated cells, positive results were reported for sister chromatid exchanges, chromosomal aberrations, and in the micronucleus test. In vivo studies in mice (Section 4.5.1.2 and Table 4-32) consistently showed genotoxic effects following dichloromethane exposure in the liver and lung, where tumors are observed. Other organs in the mouse were evaluated and mutagenic changes were not consistenly observed. The specificity of the observed effects support the hypothesized mode of action since these positive mutagenic responses are seen in organs where tumor formation occurs (i.e., liver and lung) rather than in areas that were not the site of tumors in the mouse bioassays (e.g., stomach, bladder, kidney). In vivo genotoxicity studies in rats and hamsters (the other test systems used, see Table 4-33) were predominantly non-positive. However, rats and hamsters have considerably lower GST activity than the mouse and may be less sensitive to dichloromethane-induced genotoxic effects.

In vivo binding of S-(chloromethyl)glutathione, dichloromethane's reactive GST metabolite,to DNA was not demonstrated in one study in rats and mice using a relatively low dose (5 mg/kg). The reactivity of the postulated DNA-reactive species and the instability of the derived adducts presents considerable challenges to the ability to provide direct evidence of adduct formation. Thus this lack of in vivo evidence of S-(chloromethyl)glutathione binding to DNA does not in itself represent a basis for invalidating the proposed mode of action.

4.7.3.1.1. *Experimental support for the hypothesized mode of action*

Strength, consistency, and specificity of association

It is hypothesized that mutagenic events lead to the development of liver and lung tumors following dichloromethane exposure. Several observations from experimental studies support the mutagenicity of dichloromethane and the key role of GST metabolism and the formation of DNA-reactive GST-pathway metabolites. The GST pathway produces two metabolites of dichloromethane, S-(chloromethyl)glutathione and formaldehyde, which are potentially reactive with DNA and other cell macromolecules. Enhanced dichloromethane genotoxicity in bacterial and mammalian in vitro assays with the introduction of GST metabolic capacity provides support that GST metabolism and metabolites are involved (DeMarini et al., 1997; Graves and Green, 1996; Graves et al., 1996, 1995, 1994b; Thier et al., 1993).

In bacterial strains where GST activity was not present (e.g., TA1535, TA1538), mutagenic effects were not reported following dichloromethane exposure (Gocke et al., 1981; Osterman-Golkar et al., 1983; Simula et al., 1993; Oda et al., 1996). Further tests of GST-dependent mutagenicity were evaluated by transfecting GST into non-GST bacterial strains or

decreasing GST activity in GST bacterial strains (e.g. TA100). When GSTT1-1 was cloned into bacterial strain TA1535, dichloromethane treatment resulted in reverse mutations in this new GST^+ TA1535 strain and these mutations were independent of rat S9 metabolic activation (Their et al., 1993; Pegram et al., 1997; DeMarini et al., 1997). Similarly, TA100/NG-11, a bacterial strain with decreased GST activity in comparison to the wild-type TA100 strain, showed significantly decreased mutagenicity (reverse mutations) following dichloromethane treatment (Graves et al., 1994a).

In vitro mammalian genotoxicity studies also support the importance of the GST pathway in relation to the positive effects observed following dichloromethane exposure. Positive results in the in vitro assays were limited to experiments with the presence of GST in the cell system. When mouse liver cytosol was added to hamster cell lines, dichloromethane induced DNA-protein crosslinks, DNA SSBs, and HPRT gene mutations (Graves et al., 1994b; Graves et al., 1996; Graves and Green, 1996). Additionally, in mouse Clara cells (GST is localized in the lung cells of mice), DNA SSBs were reported following dichloromethane treatment and the extent of DNA damage was significantly decreased when the cells were pretreated with a glutathione depletor (Graves et al., 1995). Other studies evaluating similar genotoxic endpoints in rat or CHO cells without modification of the low GST activity in the test system generally reported no evidence of genotoxic events (Graves et al., 1995; Andrae and Wolff, 1983; Jongen et al., 1981; Garrett and Lewis, 1983; Thilagar and Kumaroo, 1983). A study evaluating the genotoxic effects of dichloromethane (up to 6 mM) in freshly isolated mouse, rat, hamster, and human hepatocytes provides additional supporting evidence of the influence of GST activity on mutagenicity (Casanova et al., 1997). Positive results were only observed in hepatocytes from $B6C3F_1$ mice; the interspecies variability in effects correlated proportionally with the enhanced GST metabolic capacity in mice (Reitz et al., 1989). In studies with human cell lines or isolated cells, positive results were reported for sister chromatid exchanges, chromosomal aberrations, DNA damage, and in the micronucleus test. Negative results were obtained with human cells in unscheduled DNA synthesis assays (Perocco and Prodi, 1981; Jongen et al., 1981) and dichloromethane was not demonstrated to be genotoxic in studies of human hepatocytes (Graves et al., 1995; Casanova et al., 1997).

Two of three in vivo genotoxicity studies in insects reported positive results. Genotoxicity was observed in Drosophila for the gene mutation assay (Gocke et al., 1981) and the somatic assay (Rodriguez-Arnaiz, 1998) when dichloromethane was administered through the food. When Drosophila were exposed to dichloromethane via inhalation, genotoxic effects were negative as measured through gene mutation assays (sex-linked recessive lethal, somatic mutation and recombination) (Kramers et al., 1991).

In vivo genotoxicity studies reported DNA-protein cross links, DNA SSBs, chromosomal aberrations, and sister chromatid exchanges in liver cells of $B6C3F_1$ mice following acute inhalation exposure to concentrations producing liver tumors with chronic exposure (Casanova et

al., 1996, 1992; Graves et al., 1995, 1994b). The formation of DNA SSBs was suppressed when the mice were pretreated with a GSH depletor (Graves et al., 1995), providing additional support for the involvement of GST metabolism. Increased sister chromatid exchanges and chromosomal aberrations were found in the lungs in mice exposed to dichloromethane for 2 weeks to 8,000 ppm or for 12 weeks to 2,000 ppm. In this study, however, there was evidence of damage at other sites, too: sister chromatid exchanges were also seen in peripheral lymphocytes, chromosomal aberrations were seen in bone marrow and micronuclei were seen in in peripheral red blood cells under the same exposure protocol (Allen et al., 1990). As was seen in the liver, DNA SSBs were seen in lungs of $B6C3F_1$ mice following acute inhalation exposure to concentrations producing lung tumors with chronic exposure, and this effect was suppressed with pretreatment with a GSH depletor, buthionine sulfoximine (Graves et al., 1995). Other studies of sister chromatid exchange (Allen et al., 1990) or DNA damage detected by the comet assay (Sasaki et al., 1998) also provide evidence of genotoxic effects specifically in lung cells in mice. These in vivo mammaliam genotoxicity studies demonstrate site-specific effects correlating to the dichloromethane-induced tumors in animals. Additional evidence for site specificity comes from a study in which DNA damage (detected by the comet assay) was enhanced in liver tissue, but not stomach, kidney, brain, or bone marrow, 24 hours after oral administration of 1,720 mg/kg dichloromethane to CD-1 mice (Sasaki et al., 1998).

DNA reaction products (e.g., DNA adducts) produced by GST metabolites, such as S-(chloromethyl)glutathione, have not been identified in in vivo studies (Watanabet et al., 2007). The authors speculated that these results are due to the instability of the reaction products (Hashmi et al., 1994). However, adducts with nucleosides have been observed in in vitro studies, when DNA was treated with S-(1-acetoxymethyl)glutathione, a compound structurally similar to S-(chloromethyl)glutathione and in calf thymus DNA in the presence of dichloromethane, but not formaldehyde, and GST (Marsch et al., 2004; Kayser and Vuilleumier, 2001). These findings indicate that the S-(chloromethyl)glutathione intermediate formed by GSH conjugation has mutagenic potential and is likely responsible, at least in part, for the mutagenic response observed following dichloromethane exposure. However, other studies (Hu et al., 2006; Casanova et al., 1996) provide evidence of formaldehyde-related DNA-protein cross-links in relation to dichloromethane exposure. These results show that, while most studies indicate the importance of the S-(chloromethyl)glutathione intermediate in mediating genotoxic damage following dichloromethane exposure, DNA damage resulting from formaldehyde formation should also be considered.

Mutagenic data in critical genes leading to the initiation of dichloromethane-induced liver or lung tumors are not available. In vivo assays evaluating mutations in tumor suppressor genes and oncogenes reported similar frequencies of activated H-*ras* genes and inactivation of the tumor suppressor genes, *p53* and *Rb-1* in the liver tumors seen in the nonexposed and dichloromethane-exposed $B6C3F_1$ mice (Devereaux et al., 1993; Hegi et al., 1993). There were

too few lung tumors (n=4) in controls to provide a conclusive comparison of mutation patterns between exposed and nonexposed tumors.

Dose-response concordance

Statistically significant increases in liver tumor incidences in male and female (2,000 and 4,000 ppm) mice were observed in the inhalation bioassay in $B6C3F_1$ mice (NTP, 1986). Several studies provide evidence of an association between mutagenic events mediated by GST-pathway metabolites with the exposure levels inducing liver tumors in $B6C3F_1$ in this study, and concentration dependent increases in genotoxicity have been observed in in vitro and in vivo assays.

In vitro mammalian genotoxicity studies were positive and demonstrated a dose-response relationship for DNA protein crosslinks, DNA single stranded breaks, and DNA damage as measured by the comet assay at concentrations ranging from 2.5 to 60 mM when mouse liver cytosol was added or if mouse GSTT1-1 was transfected into hamster cell lines (Graves et al., 1994b; Graves et al., 1996; Hu et al., 2006). In mouse hepatocytes, DNA protein crosslinks were observed following dichloromethane exposures ranging between 0.5 – 6.0 mM (Casanova et al., 1997). DNA-protein cross-links were detected in mouse hepatocytes incubated with 1.9 mM dichloromethane (Casanova et al., 1997), a concentration chosen based on its correspondence to the TWA liver concentration of dichloromethane that was predicted by the Andersen et al. (1987) PBTK model for mice exposed by inhalation to 4,000 ppm for 6 hours (a dose that resulted in increased liver tumor incidence in the two-year bioassay reported by NTP, 1986). Consistent with the relative lack of liver tumor responses in Syrian golden hamsters (Burek et al., 1984) and F344 rats (NTP, 1986) with chronic exposure to 3,500 or 4,000 ppm, hepatocytes from these strains of animals did not form detectable DNA-protein cross-links when incubated with 1.9 mM dichloromethane (Casanova et al., 1997). .....

DNA-protein cross-links were not detected in livers of mice exposed to 146 ppm 6 hours/day for 3 days, but a concentration-dependent increase in DNA-protein cross-links was observed in DNA from livers of mice exposed to several concentrations between 500 and 4,000 ppm (Casanova et al., 1996). Following exposure under similar conditions (concentrations of 498, 1,553, or 3,923 ppm), DNA-protein cross-links were not detected in the livers of Syrian golden hamsters, a species that did not develop tumors after chronic inhalation exposure to dichloromethane (Casanova et al., 1996, 1992). Increased DNA SSBs were detected in liver tissue of $B6C3F_1$ mice immediately following a 6-hour inhalation exposure to dichloromethane at concentrations ranging from 2,000 to 8,000 ppm (Graves et al., 1995), and in mouse hepatocytes after a 3-hour exposure to 4000 (but not 2000) ppm (Graves et al., 1994b).

Statistically significant increases in the incidence of lung tumors were observed in the inhalation chronic bioassay in male and female $B6C3F_1$ mice exposed to 2,000 or 4,000 ppm dichloromethane (Mennear et al., 1988; NTP, 1986). Evidence of mutagenicity at these exposure

levels comes from two inhalation studies (Graves et al., 1995; Allen et al., 1990). Increased DNA SSBs were detected in lung tissue of $B6C3F_1$ mice immediately following a 6-hour inhalation exposure to dichloromethane at concentrations ranging from 2,000 to 8,000 ppm (Graves et al., 1995). In the study by Allen et al. (1990), increased presence of sister chromatid exchanges was observed in mouse lung cells following a 12-week exposures at 2000 ppm; shorter durations of exposure (2 weeks) were positive for measures of sister chromatid exchange and chromosome aberrations at 8000 ppm, but not at 2000 or 4000 ppm.

DNA adducts were observed and increased with dose in an in vitro preparation of calf thymus DNA when treated with dichloromethane (5 – 60 mM) and bacterial, rat, or human GST (Marsch et al., 2004).

Temporal relationship

Dichloromethane-induced liver and lung tumors first appeared in mice after 52 weeks of exposure (Maronpot et al., 1995; Kari et al., 1993). The detection of DNA-protein cross-links in the livers of $B6C3F_1$ mice following short-term inhalation exposures to dichloromethane concentrations that induced tumors with chronic exposure (Casanova et al., 1996, 1992) provides temporal support for the proposed mutagenic mode of action. Additional supporting evidence comes from observations that increased levels of DNA SSBs were detected in the liver and lungs of $B6C3F_1$ mice immediately following 3-hour inhalation exposure to 2,000–8,000 ppm dichloromethane (Graves et al., 1995; 1994b). Single dose and inhalation exposure studies of 6 hours or less did not detect an effect on DNA synthesis (Lefevre and Ashby, 1989) or unscheduled DNA synthesis (Trueman and Ashby, 1987) in mouse liver cells.

Biological plausibility and coherence

Bioactivation of a parent compound into a mutagenic metabolite resulting in cancer is a plausible mode of action carcinogenicity in humans and is a generally accepted mode of action. Dichloromethane-induced carcinogenicity is hypothesized to be due to metabolism of the parent compound by the GST pathway (GST-T1) to a metabolite that is tumorigenic. The GST metabolite, S-(chloromethyl)glutathione, formed from dichloromethane, has been characterized as labile and highly reactive through in vitro evaluation of dichloromethane metabolism in hepatocytes using ^{13}C-NMR techniques (Hashmi et al., 1994) and through an enzyme digestion assay using calf thymus DNA and GST-T1 enzyme (Marsch et al., 2004). The hypothesis that the formation of a mutagenic metabolite is a preliminary step resulting in carcinogenicity is based on evidence that malignant tumors are primarily located in areas where dichloromethane is highly metabolized by GST-T1, such as the liver and the lung, and on mutagenicity studies indicating the importance of the GST pathway and that the lung and liver are more prone to mutagenic effects of dichloromethane (Sasaki et al., 1998; Casanova et al., 1996, 1992; Graves et al., 1995, 1994b). The site selectivity of the mutagenicity in liver and lung tissue as evidenced

by several studies suggests that the GST reactive metabolite remains in the tissue where it is formed. Collectively, the studies support the hypothesis that dichloromethane-mediated carcinogenicity results from a GST metabolite that produces selective DNA damage in the tissues where the metabolite is formed, but this hypothesis is based, in part, on assumptions regarding metabolite clearance and reactivity. DNA damage in the liver and lung as well as the increased incidence of tumor formation resulting from dichloromethane exposure indicates coherence of the mutagenic and carcinogenic effects and is evidence supporting a mutagenic mode of action.

Differences in GST activity in mice compared with other species, and the interspecies variability in genotoxic effects corresponding to interspecies variability in tumor response support the mode of action hypothesis. DNA SSBs were not detected in liver or lung cells in rats exposed to similar inhalation exposures that induce strand breaks in mice (Graves et al., 1995; Graves et al., 1994b), and were detected at much lower in vitro concentrations in isolated hepatocytes from $B6C3F_1$ mice (0.4 mM) than in hepatocytes from Alpk:APfSD rats (30 mM) (Graves et al., 1995, figure 3). The difference in susceptibility to carcinogenic response between mice and rats likely reflects differences in GST metabolism. Toxicokinetic studies indicate that, with increasing exposure levels, increasing amounts of dichloromethane are metabolized via GST metabolism.

4.7.3.1.2. *Other possible modes of action for liver or lung tumors in rodents.*

Data are not available to support other possible modes of action for the liver and lung tumors in rodents. Efforts to observe sustained cell proliferation in liver following dichloromethane exposure of $B6C3F_1$ mice have been unsuccessful. Groups of female $B6C3F_1$ mice were exposed to 0 or 2,000 ppm dichloromethane 6 hours/day, 5 days/week for up to 78 weeks did not exhibit enhanced cell proliferation in the liver when assessed at various intervals during exposure (Foley et al., 1993). .

Indices of enhanced cell proliferation have been measured in the lungs of male $B6C3F_1$ mice following acute duration exposure at concentrations of about 1,500, 2,500, or 4,000 ppm dichloromethane (6 hours/day for 2 days) but not at exposure concentrations of 150 or 500 ppm and not in lungs of Syrian golden hamsters exposed to concentrations up to 4,000 ppm (Casanova et al., 1996). Earlier studies showed somewhat consistent findings in that the numbers of bronchiolar cells undergoing DNA synthesis (thymidine incorporation labeling) were markedly increased (about 6- to 15-fold) in bronchiolar cells of $B6C3F_1$ mice exposed to 4,000 ppm dichloromethane 6 hours/day on days 5, 8, and 9 of exposure, but no evidence for increased cell proliferation was found after 89, 92, or 93 days of exposure (Foster et al., 1992). The results suggest that enhanced cell proliferation is not sustained in the lung with longer-term exposure to dichloromethane concentrations associated with lung tumor development in mice,

and that this mode of tumor promotion is not important in the development of dichloromethane-induced lung tumors.

4.7.3.2. *General Conclusions About the Mode of Action for Tumors in Rodents and Relevance to Humans*

The mode of action for dichloromethane is hypothesized to involve mutagenicity via reactive metabolites. Mechanistic evidence indicates that dichloromethane-induced DNA damage in cancer target tissues of mice involves DNA-reactive metabolites produced via a metabolic pathway initially catalyzed by GST. Although mutational events in critical genes leading to tumor initiation have not been established, evidence supporting a mutagenic mode of action includes the identification of mutagenic response (reverse mutations) in short-term bacterial assays (with microsomal activation) and induced DNA-protein cross-links and DNA SSBs in mammalian cell assays. There are numerous positive in vivo genotoxicity studies specifically examining responses in the liver and/or lung; these studies included evidence of chromosomal aberrations, SSBs and sister chromatid exchanges, and DNA-protein cross-links. The negative in vivo genotoxicity assays are generally those that were based on a micronucleus test using mouse bone marrow, which is expected as halogenated hydrocarbons (such as dichloromethane) are not very effective in this type of assay (Crebelli et al., 1999; Dearfield and Moore, 2005).

Is the hypothesized mode of action sufficiently supported in test animals?

Consistent and specific evidence for the association between the formation of DNA-reactive GST-pathway metabolites and the formation of liver and lung tumors from inhalation includes (1) enhanced GST metabolic capacity in the liver and lung and enhanced localization of GST-T1 in hepatic cell nuclei in B6C3F_1 mice, compared with rats and hamsters, which do not show strong tumor responses to chronic inhalation exposure; (2) the detection of DNA-protein cross-links, or DNA SSBs in livers and lungs of B6C3F_1 mice following acute inhalation exposure to concentrations that produce tumors with chronic exposure; (3) suppression of the formation of DNA SSBs in livers and lungs of B6C3F_1 mice pretreated with a GSH depletor; (4) the inability to detect DNA-protein cross-links or DNA SSBs in livers or lungs of similarly exposed rats or hamsters (5) detection of DNA SSBs at much lower in vitro concentrations in isolated hepatocytes from B6C3F_1 mice than in hepatocytes from Alpk:APfSD rats; (6) dose-response concordance and a temporal relationship for the formation of DNA-protein cross-links and DNA SSBs with the formation of liver and lung tumors in B6C3F_1 mice exposed to dichloromethane; (7) the detection of increased sister chromatid exchanges in lung cells from CD-1 mice exposed by inhalation to dichloromethane; and (8) enhancement of dichloromethane genotoxicity in bacterial and mammalian in vitro assays with the introduction of GST metabolic

capacity. However, mutations in critical genes linked to initiation of tumor cells have not been identified.

The much weaker carcinogenic response in the liver of rats and mice to chronic drinking water exposure (Serota et al., 1986a, b) than that noted in mice exposed by inhalation (Kari et al., 1993; NTP, 1986) is correlated with much smaller amounts of GST metabolites produced in the liver under the exposure conditions of the oral bioassay than in the inhalation bioassay (Andersen et al., 1987).

In conclusion, there is sufficient evidence supporting a mutagenic mode of action and to establish the involvement of GST metabolism in the lung and liver carcinogenicity of dichloromethane in mice.

Is the hypothesized mode of action relevant to humans?

The postulated mode of action, that dichloromethane is metabolized by GST to reactive metabolites that induce mutations in DNA leading to carcinogenicity is possible in humans. Mutagenicity as a mode of action for carcinogenicity in humans is generally accepted and is a biologically plausible mechanism for tumor induction. The toxicokinetic and toxicodynamic processes that would enable reactive metabolites to produce mutations in animal models are biologically plausible in humans. Furthermore, the detection of the GST pathway in human tissues indicates that the hypothesized mode of action involving reactive metabolites from this pathway, S-(chloromethyl)glutathione and formaldehyde, is relevant to humans.

Another factor that may play a role in the apparent species differences in carcinogenicity resulting from dichloromethane exposure is species differences in intracellular localization of GST-T1 (Sherratt et al., 2002; Mainwaring et al., 1996). In mouse liver tissue, GST-T1 appears to be localized in the nuclei of hepatocytes and bile-duct epithelium, but rat liver does not show preferential nuclear localization of GST-T1. In human liver tissue, some hepatocytes show nuclear localization of GST-T1 and others show localization in cytoplasm. Nuclear production of S-(chloromethyl)glutathione catalyzed by GST-T1 in the nucleus is more likely than cytoplasmic production to lead to DNA alkylation. The finding of some nuclear localization of GST-T1 in human liver tissue supports the relevance of the hypothesized mode of action to humans.

Comparisons in mice, rats, humans, and hamsters of GST enzyme activity in liver and lung tissues have indicated the following rank order: mice > rats > or ≈ humans > hamsters (Reitz et al., 1989; Thier et al., 1998). This relative ranking in GST activity corresponds to the rank order of the strength of the association between inhalation exposure to dichloromethane and liver tumors in long-term cancer bioassays with mice, rats, and hamsters. This relative ranking does not preclude the relevance of the hypothesized mode of action to humans, however.

Which populations or lifestages can be particularly susceptible to the hypothesized mode of action?

As discussed in section 3.3, a polymorphism of the GST-T1 gene is present in humans. People with two functional copies of the gene (+/+) readily conjugate GSH to dichloromethane. Individuals having only one working copy of the gene (+/–) display relatively decreased conjugation ability. Individuals with no functional copy of the gene (–/–) do not express active GST-T1 protein and do not metabolize dichloromethane via a GST-related pathway (Thier et al., 1998). Thus the GST-$T1^{+/+}$ (wild-type) genotype would be considered to be the more "at risk" population; this subgroup represents approximately 30% of the U.S. population (Haber et al., 2002) but would be expected to be more common among Caucasians and African-Americans than among Asians (Raimondi et al., 2006; Garte et al., 2001; Nelson et al., 1995) (see Table 3-3).

According to the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA, 2005b), children exposed to carcinogens with a mutagenic mode of action are assumed to have increased early-life susceptibility. The *Supplemental Guidance* (U.S. EPA, 2005b) recommends the application of age-dependent adjustment factors (ADAFs) for carcinogens that act through a mutagenic mode of action. Although the database is lacking in vivo evidence of specific mutagenic events following chronic exposure to dichloromethane, the weight of the available evidence indicates that dichloromethane is acting through a mutagenic mode of carcinogenic action. Application of ADAFs is recommended for both the oral and inhalation routes of exposure when risks are assessed that are associated with early-life exposure.

4.8. SUSCEPTIBLE POPULATIONS AND LIFE STAGES

4.8.1. Possible Childhood Susceptibility

In humans, hepatic CYP2E1 begins to be expressed in the second trimester (Johnsrud et al., 2003), increases significantly in the third trimester, and continues to increase during the first year of life (Hines, 2007; Johnsrud et al., 2003; Treluyer et al., 1996; Vieira et al., 1996). In the fetal brain, however, CYP2E1 activity is seen as early as GD 50, with increasing levels seen until at least the end of the first trimester (Brzezinski et al., 1999). Neurobehavioral effects of dichloromethane are seen with acute exposures in adults, and the available data regarding neurological symptom history and standardized testing suggest the possibility of longer-term effects. The relatively high activity of CYP2E1 in the brain compared to the liver of the developing human fetus raises the potential for neurodevelopmental effects from dichloromethane exposure. Results from a developmental toxicity study in rats also raise concern for possible neurodevelopmental effects. Decreased offspring weight at birth and changed behavioral habituation of the offspring to novel environments were seen following exposure of adult Long-Evans rats to 4,500 ppm for 14 days prior to mating and during gestation

(or during gestation alone) (Bornschein et al., 1980; Hardin and Manson, 1980). In the only other animal study examining possible early-life susceptibility to dichloromethane toxicity, Alexeef and Kilgore (1983) found that exposure of young male mice to approximately 47,000 ppm for about 20 seconds significantly impaired the ability to learn, using a passive-avoidance conditioning task. Three-week-old mice were more affected than 5- or 8-week-old mice. The broad issue of childhood susceptibility to chronic neurobehavioral effects of early life exposure represents a data gap in the understanding of the health effects of dichloromethane.

The relatively low CYP2E1 activity in the liver of infants would tend to shift metabolism of dichloromethane to the GST pathway. This shift could affect cancer risk, given the evidence of genotoxicity through this metabolic pathway. However, the available data in humans are not sufficient to address the question of whether in utero or early life exposures represent a period of increased susceptibility to potential carcinogenic effects of dichloromethane. A threefold increased risk of childhood leukemia (acute lymphoblastic leukemia) was seen in relation to maternal occupational exposure in the year before and during pregnancy in one population-based case-control study (OR 3.22 [95% CI 0.88, 11.7]) for ratings of "probable or definite" exposure compared with possible or no exposure (Infante-Rivard et al., 2005). The estimates for categories based on concentration and frequency were similar, but there was no evidence for an increasing risk with increasing exposure level.

Experiments comparing cancer responses from early-life exposures with those from adult exposures are not available for F344 rats or $B6C3F_1$ mice, the strains of animals in which carcinogenic responses to dichloromethane have been observed (mammary gland tumors in F344 rats and liver and lung tumors in $B6C3F_1$ mice exposed by inhalation; liver tumors in female F344 rats and male $B6C3F_1$ mice exposed via drinking water). Animal data evaluating the effect of age on the susceptibility to dichloromethane carcinogenicity are restricted to a bioassay in which 54 pregnant Sprague-Dawley rats were exposed, starting on GD 12, to 100 ppm dichloromethane 4 hours/day, 5 days/week for 7 weeks, followed by 7 hours/day, 5 days/week for 97 weeks (Maltoni et al., 1988). Groups of 60 male and 69 female newborns continued to be exposed after birth to 60 ppm dichloromethane 4 hours/day, 5 days/week for 7 weeks, followed by exposure 7 hours/day, 5 days/week for 97 weeks. Additional groups of 60 male and 70 female newborns were exposed after birth to 60 ppm dichloromethane 4 hours/day, 5 days/week for 7 weeks and then for 7 hours/day, 5 days/week for 8 weeks. Endpoints monitored included clinical signs, BW, and full necropsy at sacrifice (when spontaneous death occurred). For each animal sacrificed, histopathologic examinations were performed on the following organs: brain and cerebellum, zymbal glands, interscapular brown fat, salivary glands, tongue, thymus and mediastinal lymph nodes, lungs, liver, kidneys, adrenals, spleen, pancreas, esophagus, stomach, intestine, bladder, uterus, gonads, and any other organs with gross lesions. There was no significant effect of exposure to dichloromethane on the incidence of benign or malignant tumors among adults or the progeny. The results provide no evidence that Sprague-

Dawley rats would be more sensitive to potential carcinogenic activity of dichloromethane during early life stages. Further conclusions from these results are precluded because the study included only one exposure level, which was below the maximum tolerated dose for adult Sprague-Dawley rats.

4.8.2. Possible Gender Differences

The limited data available from studies in humans do not indicate that there are large differences by gender in sensitivity to cardiovascular, neurologic, cancer, or other effects; studies have not been conducted specifically to examine this question and so do not provide information pertaining to smaller or more subtle differences. The available animal studies similarly do not establish whether either gender may be more susceptible to the toxic effects of dichloromethane. Studies of the carcinogenic effects of dichloromethane, either by inhalation or by the oral route, have not suggested an increased susceptibility of either male or female animals.

4.8.3. Other Susceptible Populations

As discussed in section 3.3, a polymorphism exists within the GST-T1 gene in humans, resulting in individuals with diminished, or a lack of, ability to conjugate GSH to dichloromethane. While the possible effects of this polymorphism on the toxicity of dichloromethane have not been directly demonstrated, it can be inferred from the proposed mode of action that a decrease in the GST-T1 metabolic pathway would result in a decreased generation of reactive metabolites and a decrease in any chronic effects mediated through those metabolites (Jonsson and Johanson, 2001; El-Masri et al., 1999).

Interindividual variation in the ability to metabolize dichloromethane via GST-T1 is associated with genetic polymorphisms in humans. Estimated U.S. population prevalence of nonconjugators (–/– at the GST-T1 locus) is about 20%, but higher prevalences (47–64%) have been reported for Asians (Raimondi et al., 2006; Haber et al., 2002; Garte et al., 2001; Nelson et al., 1995). Although nonconjugators are expected to have negligible extra risk for dichloromethane-induced cancer, the U.S. prevalences for low (+/– at the GST-T1 locus) and high (+/+) conjugators have been estimated at 48 and 32%, respectively (Haber et al., 2002). The liver and kidney are the most enriched tissues in GST-T1, but evidence is available for the presence of GST-T1 in other tissues, including the brain and lung, at lower levels (Sherratt et al., 2002, 1997).

Individuals may vary in their ability to metabolize dichloromethane through the CYP2E1 pathway. Individuals with decreased CYP2E1 activity may experience decreased generation of CO and an increased level of GST-related metabolites, following exposure to dichloromethane, which may result in increased susceptibility to the chronic effects of dichloromethane from GST-related metabolites. Conversely, individuals with higher CYP2E1 activity may experience relatively increased generation of CO at a given dichloromethane exposure level and, therefore,

may be more susceptible to the acute CO-related toxicity or other chronic effects of dichloromethane. Several studies indicate a three- to sevenfold variability in CYP2E1 activity among humans, as assessed by various types of measurements among "healthy" volunteers (Sweeney et al., 2004; Haufroid et al., 2003; Lipscomb et al., 2003; Bernauer et al., 2000; Lucas et al., 2001, 1999; Kim et al., 1995; Shimada et al., 1994). This variability is incorporated into the PBTK models for dichloromethane. Factors that may induce or inhibit CYP2E1 activity (e.g., obesity, alcohol use, diabetes) or co-exposures (i.e., to various solvents or medications) (Lucas et al., 1999) may result in greater variation within segments of the population. This variation in CYP2E1 activity may result in earlier saturation of this pathway and greater exposure to the parent compound, which would be of particular relevance to neurological effects.

5. DOSE-RESPONSE ASSESSMENTS

5.1. ORAL REFERENCE DOSE (RfD)

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

As discussed in section 4.6.1, human data for oral exposures to dichloromethane are limited to case reports involving intentional (i.e., suicidal) or accidental, acute ingestion exposures (Chang et al., 1999; Hughes and Tracey, 1993). Reported effects reflect frank toxicity from very high doses, such as marked CNS depression, injury to the gastrointestinal tract, liver and kidney failure, coma, and death. No studies of human chronic oral exposures are available. In the absence of adequate studies evaluating possible health effects in humans repeatedly exposed to dichloromethane via the oral route, the results from the chronic laboratory animal studies are assumed to be relevant to humans.

The database of laboratory animal oral exposure studies includes 90-day (Kirschman et al., 1986) and 2-year drinking water toxicity studies in F344 rats (Serota et al., 1986a) and $B6C3F_1$ mice (Serota et al., 1986b). A reproductive study exposed Charles River CD rats via gavage before mating (General Electric Co., 1976), and a developmental study exposed F344 rats via gavage during GDs 6–19 (Narotsky and Kavlock, 1995). A 14-day gavage study examined neurotoxicity in F344 rats (Moser et al., 1995).

Hepatic effects (hepatic vacuolation, nonneoplastic liver foci) are the critical dose-dependent noncancer effects associated with oral exposure to dichloromethane (see Table 4-35). The 90-day drinking water toxicity study in F344 rats (Kirschman et al., 1986) reported significant increases in hepatocyte vacuolation and necrosis in animals dosed between 166 – 1200 mg/kg-day (males) or 200 – 1469 mg/kg-day (females). These doses were used to develop dosing levels for the 104-week drinking water study (Serota et al., 1986a). The 104-week drinking water study of F344 rats (Serota et al., 1986a) provides adequate data to describe dose-response relationships for nonneoplastic liver lesions from chronic oral exposure to dichloromethane (e.g., includes four exposure levels and a control group). In this study, rats dosed at 50 mg/kg-day or higher in both sexes had increased fatty livers, but quantitative data were not provided by the authors. Liver lesions, described as foci or areas of cellular alteration, were also seen in this study in the same dose groups in which the fatty changes had occurred. A limitation of this study is that Serota et al (1986a) did not describe the evaluation of the altered foci in detail. However, increases in altered foci did not correspond to tumor rate incidences in either male or female rats. Instead, the altered foci correlated more closely to fatty liver incidence changes for both sexes in the rats. Altered foci could range from a focal fatty change (nonneoplastic) to an enzymatic altered foci change (neoplastic) (Goodman et al., 1994). Several lines of evidence were considered in determining whether the lesions should be characterized as nonneoplastic or neoplastic: 1) There is a congruence between the incidence of this lesion and

the incidence of the fatty liver in the study by Serota et al. (1986a); 2) At higher doses, hepatocyte vacuolation and hepatocyte necrosis were seen (Kirschman et al., 1986; Berman et al., 1995); and 3) there is no clear indication that these altered foci progress to liver tumors since the rate of increased foci did not correlate with liver tumor increases in either male or female rats. Based on these observations, the altered foci were determined to be more likely to be representative of a focal fatty change (nonneoplastic) than a neoplastic event.

The LOAELs for nonneoplastic liver lesions in rodents following repeated oral exposure (50–586 mg/kg-day) (Table 4-35) are in the same range or below the NOAELs of 225 mg/kg-day for reproductive performance in Charles River CD rats exposed for 90 days before mating (General Electric Co., 1976) and 450 mg/kg-day for developmental toxicity in pregnant F344 rats exposed during gestation (Narotsky and Kavlock, 1995). The LOAEL (337 mg/kg-day) and NOAEL (101 mg/kg-day) for mild neurological impairment in a 14-day gavage exposure study of F344 rats (Moser et al., 1995) indicates that the threshold for neurological effects may be similar to the threshold for liver effects. A limitation of the Moser et al. (1995) study, however, is that the observed effects were limited to measures taken within 4 hours of exposure.

The subchronic (i.e., 90-day or less study) data were not considered in the selection of a principal study for deriving the chronic RfD because the database contains reliable dose-response data from a chronic study at lower doses than the 90-day study (Kirschman et al., 1986) (conducted to provide data pertaining to relevant doses to use in the chronic study). The data from the subchronic studies are, however, used to corroborate the findings in the chronic studies with respect to relevant endpoints (i.e., hepatic and neurological effects). The neurotoxicity study was not selected as the principal study due to the limited measurements to inform the chronic exposure to dichloromethane. The rat rather than the mouse chronic bioassay (Serota et al., 1986a) was selected as the principal study for the RfD because of the consistent evidence that rats may be more sensitive than mice to nonneoplastic liver effects from orally administered dichloromethane; available rat LOAELs for nonneoplastic liver lesions are lower than mouse LOAELs (see Table 4-35). Figure 5-1 is an exposure-response array that presents NOAELs, LOAELs, and the dose range tested, corresponding to selected health effects from the short-term (neurotoxicological) and subchronic studies, and from the chronic, reproductive, and developmental toxicity studies that were evaluated for use in the derivation of the RfD.

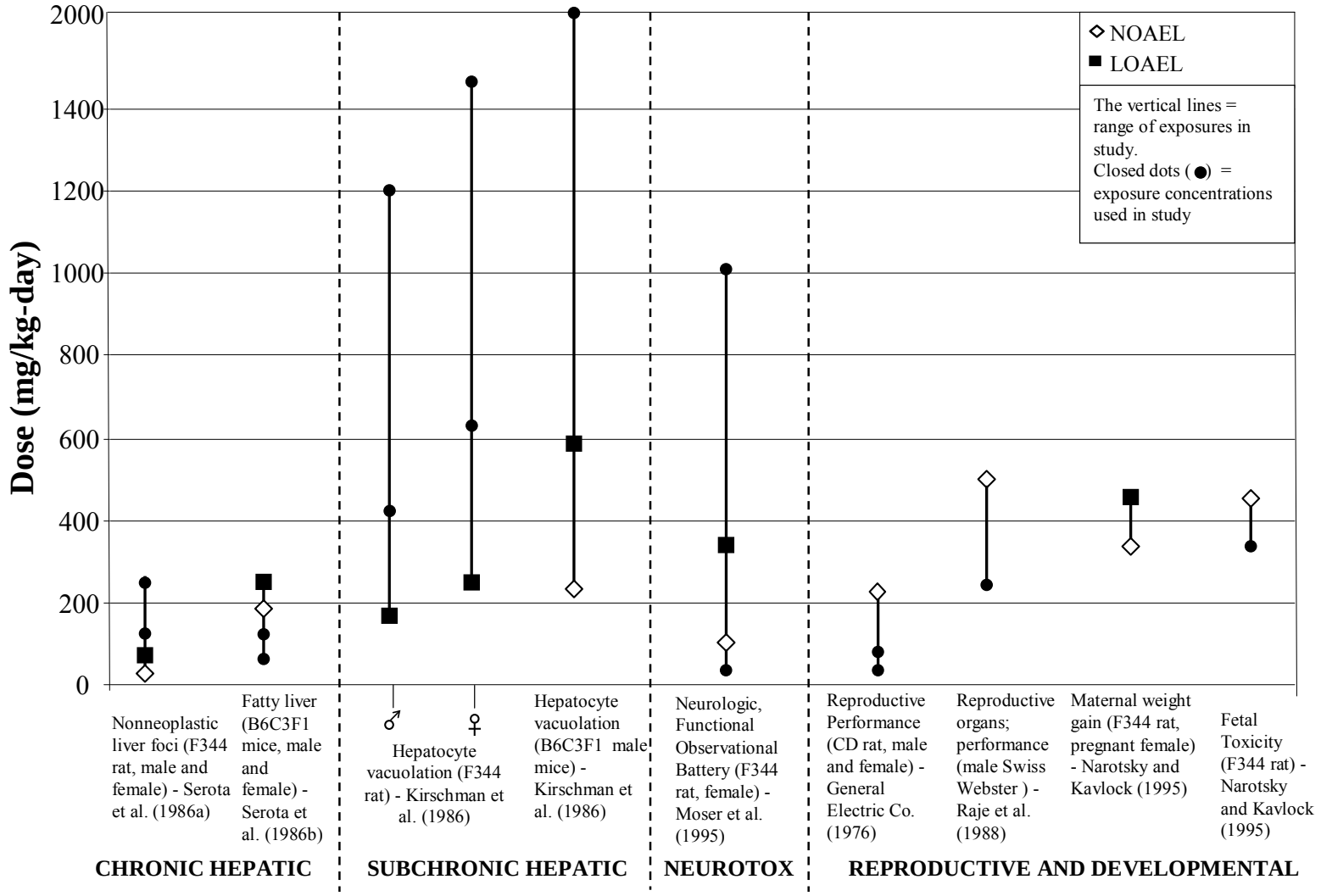

Figure 5-1. Exposure Response Array for Oral Exposure to Dichloromethane.

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6676 **Figure 5-1. Exposure response array for oral exposure to dichloromethane**

5.1.2. Derivation Process for Noncancer Reference Values

The toxicity values (oral RfD and inhalation RfC) for noncancer endpoints were derived by using rat and human PBTK models to calculate internal doses in rats from experimental exposures and extrapolate points of departure to human equivalent exposures. Figure 5-2 illustrates the process of using the PBTK models for toxicity value derivation. The process for the RfD and RfC is summarized below, using the example of a noncancer liver effect.

A deterministic PBTK model for dichloromethane in rats was first used to convert rat drinking water or inhalation exposures to values of an internal liver dose metric (see Appendix C for details of the rat PBTK model). Available models in EPA benchmark dose (BMD) software (BMDS) version 2.0 were then fit to the liver lesion incidence data and internal liver dose data for rats, and BMD_{10}s and their lower 95% confidence limits associated with a 10% extra risk ($BMDL_{10}$) were calculated from each of the models. Adequacy of model fit was assessed by overall χ^2 goodness of fit (p-value > 0.10) and examination of residuals, particularly in the region of the benchmark response (BMR). The choice of best-fitting model was based on the lowest Akaike's Information Criterion [AIC] among models with adequate fits (U.S. EPA, 2000b).

The use of a PBTK model can replace the use of the $BW^{0.75}$ scaling factor to account for interspecies differences in toxicokinetics. The decision with respect to use of a scaling factor depends on the dose metric that is used. Where PBTK models predict the concentration (in particular, the AUC) of the proximate causative agent, a scaling factor to account for interspecies differences is not typically used. That is, it is assumed that if the time-averaged (or steady-state) concentration of the proximate causative agent predicted by the PBTK model in the target tissue is the same in the test species as in humans, and the test species was exposed for an equivalent portion of its lifetime (2 years in rats and mice being equivalent to a 70-year lifetime in humans), then the resulting risks in the two species are the same. However, when the PBTK model predicts the rate of production of the agent, rather than its concentration, then a $BW^{0.75}$ scaling factor may be appropriate, depending on what is known or expected regarding the rate of clearance of the agent or metabolite of interest. Two different scenarios can be considered. If the metabolite formed is considered to be highly reactive, then it can be assumed that the rate of clearance (i.e., disappearance due to local reactivity) for this metabolite, per volume tissue, is equal in rodents and humans. Thus, in that situation, as with the AUC dose metric, no $BW^{0.75}$ scaling factor is necessary, although differences in tissue volume fraction in humans versus rats (as occurs for liver) should be and are accounted for by the PBTK model. However, if the metabolite is not highly reactive, then it is expected that interspecies differences in clearance or removal of the toxic metabolite follow the generally assumed $BW^{0.75}$ scaling for rates of metabolism and blood circulation. In this case, or in situations in which the reactivity or rate of removal of the metabolite has not been established, it is appropriate to use a scaling factor, based on BW ratios, to account for this difference. In the case of the noncancer liver effects of dichloromethane, very limited information is available on the mechanism(s) involved in creating

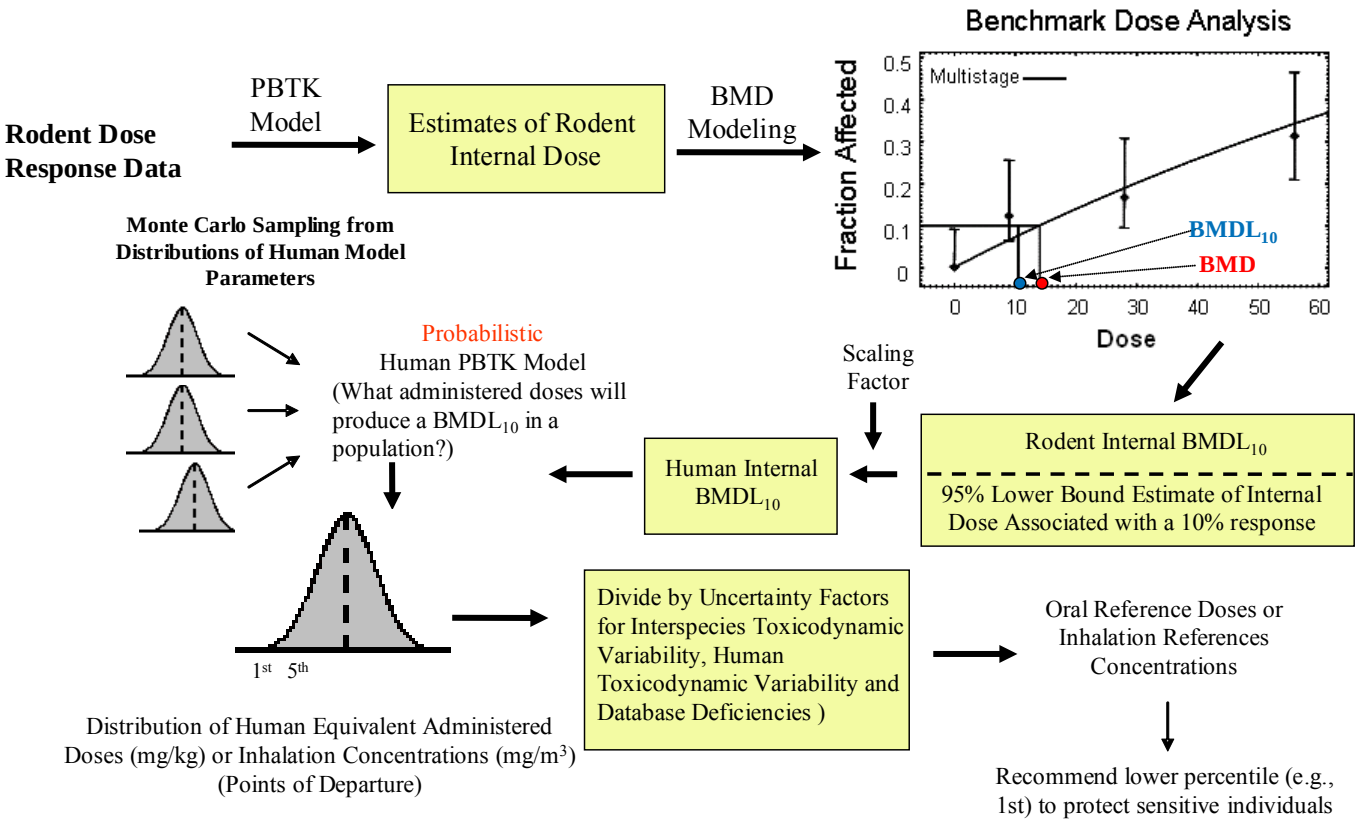

Figure 5-2. Process for deriving noncancer oral RfDs and inhalation RfCs using rodent and human PBTK models.

the type of hepatic damage seen. The dose metric used in the PBTK modeling is a rate of metabolism, rather than the concentration of putative toxic metabolites, and the clearance of these metabolites may be slower per volume tissue in the human compared with the rat. Thus the rat internal dose metric for noncancer effects was adjusted by dividing by a pharmacokinetic scaling factor to obtain a human-equivalent internal $BMDL_{10}$

A probabilistic PBTK model for dichloromethane in humans, adapted from the model of David et al. (2006) as described in Appendix B, was then used to calculate distributions of chronic exposures associated with the human equivalent internal $BMDL_{10}$, based on the responses in rats. Parameters in the human PBTK model are distributions that incorporate information about dichloromethane toxicokinetic and physiological variability and uncertainty among humans, incorporating information about both the CYP2E1 and GST-T1 metabolic pathways (see Table 3-9 and Appendix B). Monte Carlo sampling was performed in which each human model parameter was defined by a value randomly drawn from each respective parameter distribution. The model was then executed by using the human internal $BMDL_{10}$ as input, and the resulting human equivalent administered dose or human equivalent concentration (HEC) was recorded. This process was repeated for 10,000 iterations to generate a distribution of human equivalent administered doses or concentrations.

The parameter statistics reported by David et al. (2006) include both the inter-individual variability that would have been elucidated by the Bayesian analysis (variation between mean values for each individual for which data were available) and uncertainty in those values. Since EPA's objective is to account for both population variability and parameter uncertainty, however, these statistics were primarily used as published in David et al. (2006) (exceptions discussed in Appendix B) to define population distributions. Assuming that these parameters are distributed independently, ignoring the covariance that was likely represented in the actual posterior chains, will tend to over-estimate the overall range of parameters and hence distribution of dose metrics in the population, compared to what one would obtain if the covariance were explicitly included. Thus if the covariance (i.e., the variance-covariance matrix) for the set of parameters had been reported by David et al., it could have been used to narrow the predicted distribution of internal doses, or equivalent applied doses. Lacking such information the approach used will not under-estimate risk or over-estimate lower bounds on human equivalent exposure levels.

From these distributions of human equivalent administered doses (or concentrations), candidate RfDs or RfCs were derived by dividing the first percentile value (point of departure) by uncertainty factors (UFs) to account for uncertainty about potential interspecies toxicodynamic variability, human toxicodynamic variability, and database deficiencies. The first percentile was chosen because it allowed generation of a stable estimate for the lower end of the distribution while being protective of the overall human population, including

sensitive individuals. Choosing this lower point replaces the use of an additional UF to account for human toxicokinetic variability.

5.1.3. Evaluation of Dose Metrics for Use in Noncancer Reference Value Derivations

There are no data to support the role of a specific metabolite in the development of the noncancer liver lesions seen in oral and inhalation exposure studies. Four dose metrics were examined as potential metrics for the internal dose of interest: rate of hepatic metabolism through the CYP pathway, rate of hepatic metabolism through the GST pathway, the combined rate of hepatic metabolism through the CYP and GST pathways, and the concentration (area under the curve, AUC) of dichloromethane, the parent compound, in the liver. The dose-response patterns for each of these metrics in the oral study in rats (Serota et al., 1986a) and in two inhalation studies in rats (Nitscke et al., 1988a; Burek et al., 1984) were examined for fit and congruence.

Using the oral exposure data, only one of the seven models, the log-logistic model, produced an adequately fit ($p > 0.10$) for the GST metabolism metric and the dichloromethane AUC metrics. Adequate model fit was seen in all of the models using the CYP dose metric with the oral data, and using the GST, CYP, and AUC dose metrics for the inhalation data.

A limitation in using the GST metric can be observed when comparing the oral and inhalation responses at various exposure levels. At 200 ppm, where the GST metric is predicted by the PBTK model to be 93 mg metabolism/L liver/day, no liver effects were seen. In contrast, liver responses were elevated at an oral dose of 50 mg/kg-day, where the GST metric is predicted to be 60 mg metabolism/L liver/day (see Tables 5-1 and 5-5, respectively, for the oral and inhalation internal metrics). Thus the liver GST metric produces an inconsistency in the dose-response relationship, with very different responses observed depending on the route of exposure. A similar inconsistency occurs with the AUC metric. These differences are not observed, however, when using the CYP metric. At the 200 ppm inhalation exposure, where no hepatoxicity was observed, the CYP metric is predicted to be 660 mg/L liver/day. This internal CYP metabolism metric is less than that predicted for the oral dose for the 50 mg/kg-day group (i.e., 872 mg metabolism/L liver/day), in which liver effects were observed. Thus, the CYP internal metric is consistent with the observed responses seen in the oral and inhalation exposure studies.

The GST metabolism and the AUC dose metrics did not present reasonable choices based on model fit and consistency of response across studies at comparable dose levels. Given these results, the combination of hepatic metabolism through the GST and the CYP pathways would not be expected to result in an improvement to a metric based only on CYP metabolism. The CYP-metabolism dose metric is the most consistent with the data. This metric was selected for the subsequent RfD and RfC derivations. The lack of information on

mechanisms with respect to noncancer health effects represents data gaps in the understanding of the health effects of dichloromethane.

5.1.4. Methods of Analysis—Including Models (PBTK, BMD, etc.)

PBTK models for dichloromethane in rats were described previously in section 3.5. From the evaluation described in Appendix C, a modified model of Andersen et al. (1991) was selected for the calculation of internal dosimetry of ingested dichloromethane in the rats in the principal study (Serota et al., 1986a).

PBTK model simulations of the drinking water study of Serota et al. (1986a) (Table 5-1) were performed to calculate average lifetime daily internal liver doses in male and female F344 rats. In the absence of data for group- and sex-specific BWs, reference values were used for male and female F344 rats in chronic studies (U.S. EPA, 1988a). The mode of action by which dichloromethane induces nonneoplastic liver effects in rodents has not received research attention to determine the role of the parent material, metabolites of the CYP2E1 pathway, metabolites of the GST pathway, or some combination of parent material and metabolites. In the absence of this kind of knowledge, and considering the pattern of response seen in the oral and inhalation studies (as described in section 5.1.3.), an internal dose metric based on the amount of dichloromethane metabolized via the CYP pathway in the liver (mg dichloromethane metabolized via CYP pathway per liter liver per day) was used. Figure 5-3 shows the comparison between oral external and internal doses, using this dose metric for the rat and for the human.

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Table 5-1. Incidence data for nonneoplastic liver lesions and internal liver doses, based on various metrics, in male and female F344 rats exposed to dichloromethane in drinking water for 2 years (Serota et al., 1986a)

Sex	Nominal (actual) daily intake (mg/kg-day)	Rat liver lesion incidence[a]	Rat internal liver dose[b]			
			CYP	GST	GST and CYP	Parent AUC
Male (BW = 380 g)	0 (0)	52/76 (68%)	0	0	0	0
	5 (6)	22/34 (65%)	133.9	2.1	136.1	0.5
	50 (52)	35/38 (92%)[c]	872.7	58.8	931.4	13.1
	125 (125)	34/35 (97%)[c]	1,433.1	236.0	1,669.1	52.6
	250 (235)	40/41 (98%)[c]	1,868.6	561.5	2,430.0	125.0
Female (BW = 229 g)	0 (0)	34/67 (51%)	0	0	0	0
	5 (6)	12/29 (41%)	134.5	2.1	136.6	0.4
	50 (58)	30/41 (73%)[c]	977.8	66.0	1,043.8	12.6
	125 (136)	34/38 (89%)[c]	1577.0	258.7	1,835.7	49.5
	250 (263)	31/34 (91%)[c]	2070.0	642.4	2,712.3	122.9

[a]Liver foci/areas of cellular alteration; number affected divided by total sample size.
[b]Internal doses were estimated using a rat PBTK model from simulations of actual daily doses reported by the study authors. CYP dose is in units of mg dichloromethane metabolized via CYP pathway/L tissue/day; GST dose is in units of mg dichloromethane metabolized via GST pathway/L tissue/day.; GST and CYP dose is in units of mg dichloromethane metabolized via CYP and GST pathways/L tissue/day; and Parent AUC dose is in units of mg dichloromethane*hrs)/L tissue.
[c]Significantly ($p < 0.05$) different from control with Fisher's exact test.

Source: Serota et al., 1986a.

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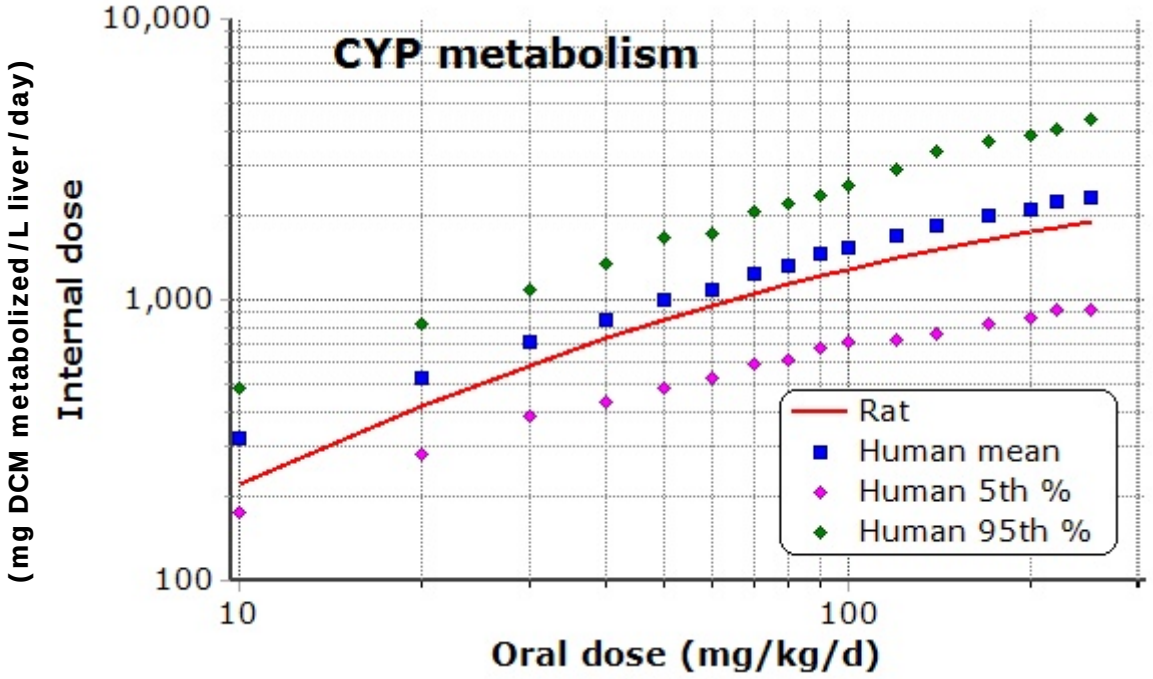

Figure 5-3. PBTK model-derived internal doses (mg dichloromethane metabolized via the CYP pathway per liter liver per day) in rats and humans and their associated external exposures (mg/kg-day), used for the derivation of RfDs. Six simulated daily drinking water episodes are described by Reitz et al. (1997). The human metabolism rates were estimated using a computational sample of 1000 individuals per dose, including random samples of the three GST-T1 polymorphisms (+/+, +/-, -/-) in the current U.S. population based on data from Haber et al. (2002). Since a different set of samples was used for each dose, some stochasticity is evident as the human points (values) do not fall on smooth curves.

The seven dichotomous dose-response models in BMDS version 2.0 were fit to the rat liver lesion incidence data and PBTK model-derived internal dose data to derive a rat internal BMD_{10}, and corresponding $BMDL_{10}$, associated with 10% extra risk (Table 5-2). The quantal model is identical to the one-stage multistage model and so is not included in this set of models. A BMR of 10% was selected because, in the absence of information regarding the magnitude of change in a response that is thought to be minimally biologically significant, a BMR of 10% is generally recommended, since it provides a consistent basis of comparison across assessments. There are no additional data to suggest that the critical response has a greater sensitivity that would warrant a lower BMR. The male rats exhibited a greater sensitivity compared to the female rats (based on lower $BMDL_{10}$ values for all of the models examined) and thus the male data are used as the basis for the RfD derivation. The logistic model was the best fitting model for the male incidence data, based on AIC value among models with adequate fit (U.S. EPA,2000b). Modeling results are shown in detail in Appendix D-1).

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Table 5-2. BMD modeling results for incidence of noncancer liver lesions in male and female F344 rats exposed to dichloromethane in drinking water for 2 years, based liver-specific CYP metabolism dose metric (mg dichloromethane metabolism via CYP pathway per liter liver tissue per day)

Sex and model[a]	BMD_{10}	$BMDL_{10}$	χ^2 goodness of fit *p*-value	AIC
Males				
Gamma[a]	151.73	48.93	0.62	185.33
Logistic	**85.17**	**61.78**	**0.75**	**183.61**
Log-logistic[a]	213.73	37.06	0.83	184.79
Multistage (1)[a]	68.62	47.58	0.71	183.74
Probit	98.87	75.49	0.69	183.81
Log-probit[a]	197.65	77.56	0.81	184.84
Weibull[a]	117.29	48.39	0.57	185.49
Females				
Gamma[a]	336.38	98.70	0.52	233.07
Logistic	169.77	134.87	0.59	231.70
Log-logistic[a]	404.87	101.15	0.60	232.80
Multistage (1)[a]	123.59	91.46	0.47	232.32
Probit	179.59	146.27	0.59	231.70
Log-probit[a]	400.95	173.57	0.60	232.80
Weibull[a]	283.24	97.31	0.47	233.27

[a]These models in EPA BMDS version 2.0 were fit to the rat dose-response data shown in Table 5-1 by using internal dose metrics calculated with the rat PBTK model. Details of the models are as follows: Gamma and Weibull models restrict power ≥1; Log-logistic and Log-probit models restrict to slope >1, multistage model restrict betas ≥0; lowest degree polynomial with an adequate fit is reported (degree of polynomial noted in parentheses).
Bolded model is the best-fitting model in the most sensitive sex (males), which is used in the RfD derivation.

Source: Serota et al. (1986a).

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6848 The $BMDL_{10}$ from the logistic model was used as the point of departure for the RfD
6849 calculations (Table 5-3). This rat internal dose metric for noncancer effects was adjusted to
6850 obtain a human-equivalent internal $BMDL_{10}$ by dividing by a pharmacokinetic scaling factor
6851 based on a ratio of BWs $(BW_{human}/BW_{rat})^{0.25} = 4.09$). This scaling factor was used because the
6852 metric is a rate of metabolism, rather than the concentration of putative toxic metabolites, and the
6853 clearance of these metabolites may be slower per volume tissue in the human compared with the
6854 rat (that is, total rate of removal may scale as $BW^{0.75}$, while tissue volume scales as BW^1).
6855 The human PBTK model (adapted from David et al. [2006], as described in Appendix B),
6856 using Monte Carlo sampling techniques, was used to calculate quantiles of human equivalent
6857 administered oral daily doses (in mg/kg-day) associated with the internal $BMDL_{10}$ values

(Table 5-3), as described above in section 5.1.2. The human model used parameter values derived from Monte Carlo sampling of probability distributions for each parameter, including MCMC-derived distributions for the metabolic parameters for the metabolism through the CYP2E1 pathway (V_{max} and K_m) and a distribution of GST metabolic rate constants that is weighted to reflect the estimated frequency of GST-T1 genotypes (20% GST-$T1^{-/-}$, 48% GST-$T1^{+/-}$, and 32% GST-$T1^{+/+}$) in the current U.S. population, based on data from Haber et al. (2002). All simulations also included a distribution of CYP activity based on data from Lipscomb et al. (2003). The drinking water exposures were comprised of six discrete drinking water episodes for specified times and percentages of total daily intake (Reitz et al., 1997). The mean and two lower points on the distributions of human equivalent administered daily doses derived from the Serota et al. (1986a) data for male rats, using the $BMDL_{10}$ from the logistic model , are shown in Table 5-3.

Table 5-3. RfD for dichloromethane based on PBTK model-derived probability distributions of human drinking water exposures extrapolated from nonneoplastic liver lesion incidence data for male rats exposed via drinking water for 2 years, based on liver-specific CYP metabolism dose metric (mg dichloromethane metabolized via CYP pathway per liter liver tissue per day)

Model[a]	Rat internal $BMDL_{10}$[b]	Human internal $BMDL_{10}$[c]	Human equivalent dose (mg/kg-day)[d]			Human RfD (mg/m^3)[e]
			1st percentile	5th percentile	Mean	
Logistic	61.78	15.11	0.214	0.252	0.395	7×10^{-3}

[a]Based on the best-fitting model from Table 5-2.
[b]Rat dichloromethane PBTK model-derived internal liver dose associated with the lower bound on 10% extra risk for developing liver foci/areas of cellular alteration.
[c]Human dichloromethane internal liver dose, derived by dividing the rat internal $BMDL_{10}$ by a scaling factor of 4.09 $[(BW_{human}/BW_{rat})^{0.25}]$ to account for potential interspecies pharmacokinetic differences in the clearance of metabolites.
[d]PBTK model-derived distributions of daily average dichloromethane drinking water doses predicted by the PBTK model to yield an internal dose in humans equal to the dichloromethane internal $BMDL_{10}$,
[e]Human RfD, based on male rat data, derived by dividing the 1st percentile of human equivalent dose value by a total UF of 30: 3 ($10^{0.5}$) for possible toxicodynamic differences between species, 3 ($10^{0.5}$) for variability in human toxicodynamic response, and 3 ($10^{0.5}$) for database deficiencies. The 1st percentile point of departure is a stable estimate of the lower end of the distribution. Use of this value in the lower tail replaces use of a UF for human toxicokinetic variability.

Source: Serota et al. (1986a).

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

The 1st percentile point of departure is a stable estimate of the lower end of the distribution. Use of this value associated with a sensitive human population addresses the uncertainty associated with human toxicokinetic variability. To derive the candidate RfD based on data from male rats, the first percentile value of the distribution of human equivalent administered dose associated with the male rat-derived $BMDL_{10}$ was divided by a composite UF of 30 (3 [$10^{0.5}$] to account for uncertainty about interspecies toxicodynamic equivalence, 3 [$10^{0.5}$] to account for uncertainty about toxicodynamic variability in humans, and 3 [$10^{0.5}$] for database deficiencies) (Table 5-3). The resulting RfD recommended for dichloromethane is 7×10^{-3} mg/kg-day.

In deriving this RfD, factors for the following areas of uncertainty were considered:

- *Uncertainty in extrapolating from laboratory animals to humans (UF_A).* The use of PBTK models to extrapolate internal doses from rats to humans reduces toxicokinetic uncertainty in extrapolating from the rat liver lesion data but does not account for the possibility that humans may be more sensitive than rats to dichloromethane due to toxicodynamic differences. A UF of 3 ($10^{0.5}$) to account for this toxicodynamic uncertainty was used, as shown in Table 5-3.

- *Uncertainty about variation from average humans to sensitive humans (UF_H).* The probabilistic human PBTK model used in this assessment incorporates the best available information about variability in toxicokinetic disposition of dichloromethane in humans but does not account for humans who may be sensitive due to toxicodynamic factors. Thus, a UF of 3 ($10^{0.5}$) was applied to account for possible toxicodynamic differences in sensitive humans.

- *Uncertainty in extrapolating from LOAELs to NOAELs (UF_L).* A UF for extrapolation from a LOAEL to a NOAEL was not applied because BMD modeling was used to determine the POD, and this factor was addressed as one of the considerations in selecting the BMR. The BMR was selected based on the assumption that it represents a minimum biologically significant change.

- *Uncertainty in extrapolating from subchronic to chronic durations (UF_S).* The derived RfD is based on results from a chronic-duration drinking water toxicity study. No cross-duration UF is necessary.

- *Uncertainty reflecting incompleteness of the overall database (UF_D).* The oral database for dichloromethane includes well-conducted lifetime drinking water studies in rats

(Serota et al., 1986a) and mice (Serota et al., 1986b) and a supporting subchronic study in rats and mice (Kirschman et al., 1986). These studies provided dose-response data for the hepatic effects of dichloromethane. The database also includes one-generation oral reproductive toxicity (General Electric Co., 1976) and developmental toxicity (Narotsky and Kavlock, 1995) studies that found no reproductive or developmental effects at dose levels in the range of doses associated with liver lesions. A two-generation oral exposure study is not available; however, a two-generation inhalation exposure study by Nitschke et al. (1988a) reported no effect on fertility index, litter size, neonatal survival, growth rates, or histopathologic lesions at exposures of ≥100 ppm. However, there have been no oral exposure studies that evaluated neurobehavioral effects in offspring. This is a relevant endpoint given the increase in blood CO (a known developmental neurotoxicant) that occurs through the CYP2E1 metabolic pathway for dichloromethane after oral and inhalation exposures. There are no oral exposure studies that include functional immune assays; however, there is a 4-week inhalation study of potential systemic immunotoxicity that found no effect of dichloromethane exposure at concentrations up to 5,000 ppm on the antibody response to sheep red blood cells (Warbrick et al., 2003). The Warbrick et al. (2003) data suggest that systemic immunosuppression is not a concern for dichloromethane exposure. Route-specific local immunosuppression from acute inhalation exposure in CD1 mice was seen in Aryani et al. (1986). The findings from Aryani et al. (1986) were considered to be portal-of-entry effects involving local immunosuppression within the lung (Streptococcus and Klebsiella infectivity models) and unlikely to be observed following oral exposure. Because of concern regarding the adequacy of available data pertaining to possible neurodevelopmental toxicity and the lack of a two-generation reproductive study, a UF_D of 3 was applied.

5.1.6. *Previous RfD Assessment*

The previous IRIS assessment derived an RfD of 0.06 mg/kg-day based on the NOAELs of 5.85 and 6.47 mg/kg-day for nonneoplastic liver toxicity (foci/areas of cellular alteration) in male and female rats, respectively, in a 2-year drinking water study (Serota et al., 1986a). The LOAELs associated with these NOAELs were 52.58 and 58.32 mg/kg-day for males and females, respectively. The RfD of 0.06 mg/kg-day was derived by dividing the average NOAEL of 6 mg/kg-day (for male and female rats) by a UF of 100 (10 for intraspecies variability and 10 for interspecies variability).

5.1.7. *RfD Comparison Information*

Use of the mean value (3.95×10^{-1} mg/kg-day) of the human equivalent administered dose distribution instead of the 1st percentile, with an additional UF of 3 ($10^{0.5}$) to account for human toxicokinetic variability, would yield an RfD of 4×10^{-3} mg/kg-day.

Additional comparisons between the derived RfD and values developed from other endpoints or data sets using NOAEL/LOAEL methods are shown in Table 5-4 and Figure 5-4. NOAELs were used as comparison points of departure and were not scaled allometrically. The point of departure for three endpoints (Serota et al., 1986a; Moser et al., 1995; Narotsky and Kavlock) are presented in Table 5-4. For Serota et al. (1986a), this is based on BMD modeling of a 10% increase in liver lesions using internal liver dose metric (mg dichloromethane metabolism via CYP pathway per liter liver tissue per day) derived from a rat PBTK model. After an allometric scaling factor of 4.09 was applied, the human internal $BMDL_{10}$ was 15.11 mg/kg-day. A probabilistic human PBTK model adapted from David et al. (2006) was used to generate a distribution of human equivalent doses from the human internal $BMDL_{10}$ and the first percentile of this distribution was used as the point of departure. For other studies, POD is based on the lowest non-control dose in which no effect was seen. These NOAELs that were used as points of departure and were not scaled allometrically

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Table 5-4. Potential points of departure with applied UFs and resulting candidate RfDs

Endpoint	POD[a] (mg/kg-day)	POD Type and Description	Uncertainty Factors Applied[b]						Candidate RfD (mg/kg-day)	Reference
			Total UF	UF_A	UF_H	UF_L	UF_S	UF_D		
Nonneoplastic liver foci, male rats	**61.78**	**BMD; 10% increase in incidence of liver lesion**	**30**	**3**	**3**	**1**	**1**	**3**	**7×10^{-3}**	**Serota et al. (1986a)**
Neurological changes (FOB), female rats	101	NOAEL; No effect at POD, approximate doubling of severity score of neuromuscular and sensorimotor domains	3,000	10	10	1	10	3	3.4×10^{-2}	Moser et al. (1995)
Maternal weight gain, female rats	338	NOAEL; No effect at POD, approximate 33% decrease in weight gain seen at next dose	300	10	10	1	1	3	1.1	Narotsky and Kavlock (1995)

[a]POD = point of departure. .

[b]UF_A = uncertainty in extrapolating from laboratory animals to humans, UF_H = uncertainty about variation from average humans to sensitive humans, UF_L = uncertainty about extrapolating from LOAEL to NOAEL, UF_S = uncertainty in extrapolating from subchronic to chronic durations, and UF_D = uncertainty reflecting incompleteness of the overall database. A UF for extrapolation from a LOAEL to NOAEL (UF_L) was not used for any of these studies. For the Serota et al., (1986a) study, the use of the first percentile of the human equivalent dose distribution as the point of departure replaces the use of a UF_H for human toxicokinetic variability.

Bolded value is the basis for the RfD of 7×10^{-3} mg/kg-day.

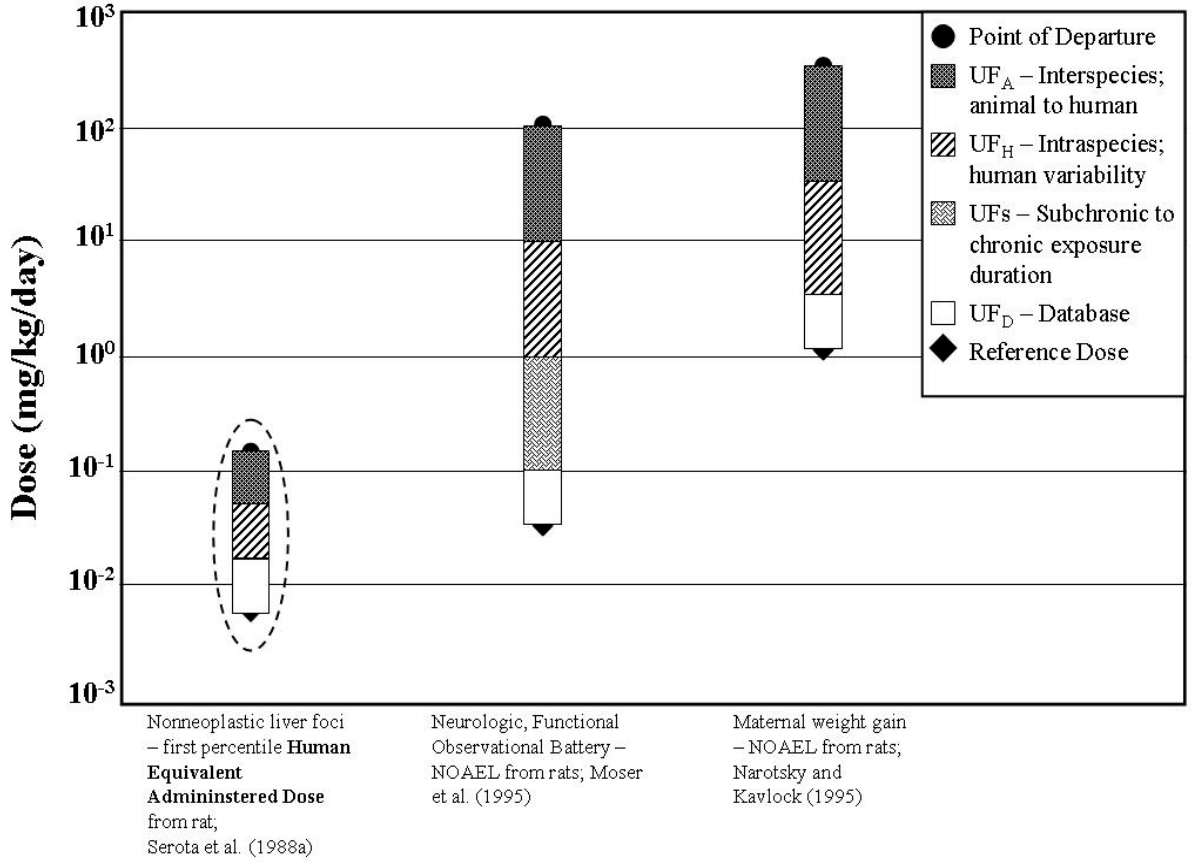


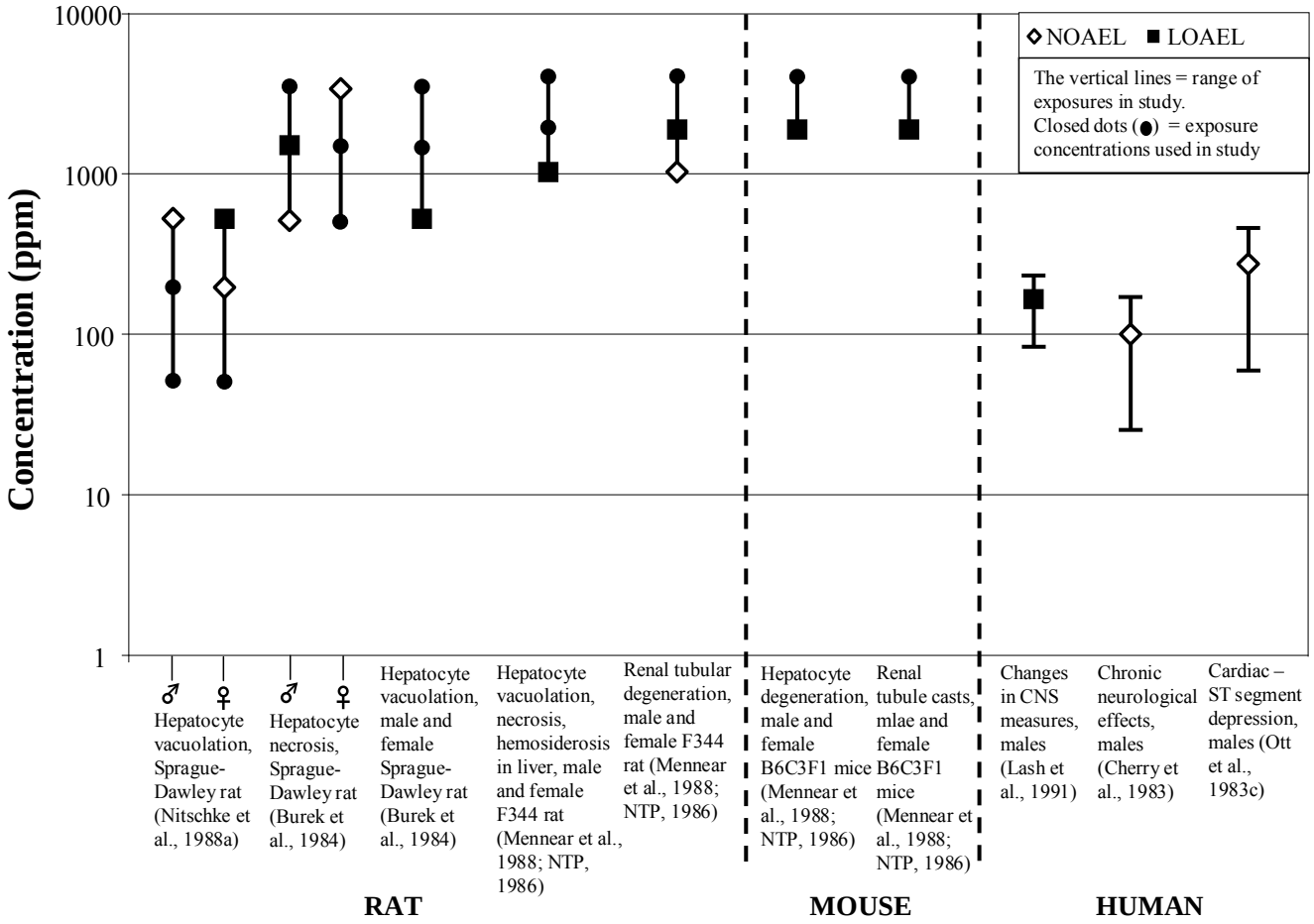
Figure 5-4. Comparison of candidate RfDs derived from selected point of departures for endpoints presented in Table 5-4.

5.2. INHALATION REFERENCE CONCENTRATION (RfC)

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

Figure 5-5 includes exposure-response arrays from some of the human studies that were evaluated for use in the derivation of the RfC. Several acute-duration controlled exposure studies (section 4.1.2.2) and cross-sectional occupational studies (sections 4.1.2.3 and 4.1.2.4) in humans are available that show neurological effects from dichloromethane exposure. These effects include an increase in prevalence of neurological symptoms among workers (Cherry et al., 1981) and possible detriments in attention and reaction time in complex tasks among retired workers (Lash et al., 1991). However, these studies have inadequate power for the detection of effects with an acceptable level of precision. In addition, the Cherry et al. (1981) study is limited by the definition and documentation of neurological symptom history, and the Lash et al. (1991) study has exposure measurements from 1974–1986, but the work histories of exposed workers go back to the 1940s. Ott et al. (1983c) reported an increase in serum bilirubin among exposed workers, but there was no association seen with respect to the other hepatic enzymes examined (serum γ-glutamyl transferase, serum AST, serum ALT), and no evidence of hepatic effects was seen in a later study of the same cohort (Soden, 1993). Because of these limitations, these human studies of chronic exposures do not serve as an adequate basis for RfC derivation. As discussed in section 5.2.6, however, the quantitative measures of neurological function from Cherry et al. (1983) were used to derive a comparative RfC.

The database of experimental animal dichloromethane inhalation studies includes numerous 90-day and 2-year studies, with data on hepatic, pulmonary, and neurological effects, (see Table 4-36) and reproductive and developmental studies (Table 4-37) (see summary in Section 4.6.2). NOAELs, LOAELs, and the dose range tested corresponding to selected health effects from the chronic studies are shown in Figure 5-5, and effects seen in subchronic, reproductive, and developmental studies are shown in Figure 5-6. The subchronic (i.e., 90-day or less study) data were not considered in the selection of a principal study for deriving the RfC because the database contains reliable dose-response data from the chronic study at lower doses than the 90-day study. The data from the subchronic studies are, however, used to corroborate the findings with respect to relevant endpoints (i.e., hepatic and neurological effects).



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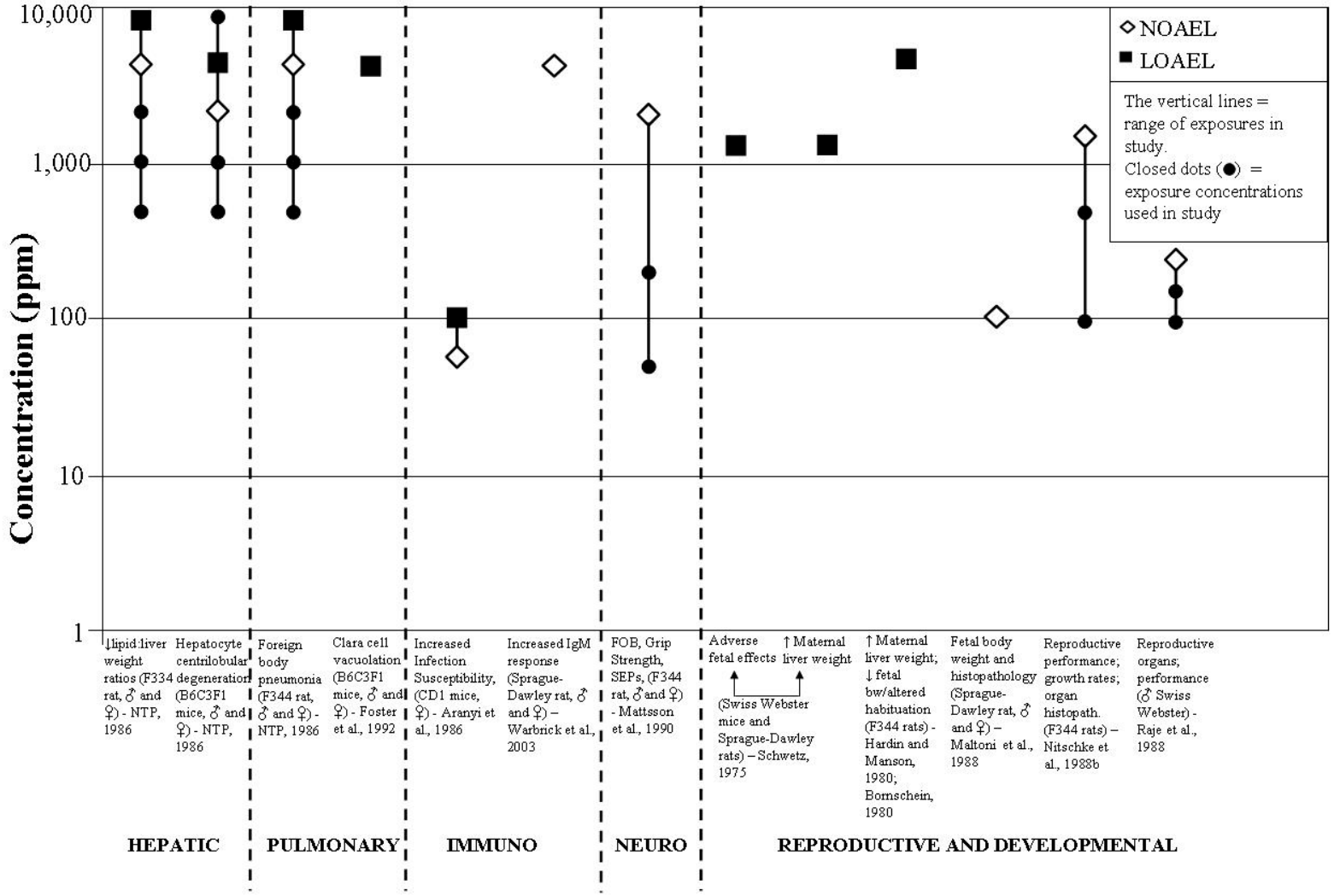
6999 **Figure 5-5. Exposure response array for chronic (animal) or occupational (human) inhalation exposure to**
7000 **dichloromethane (log Y axis)**



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7003 **Figure 5-6. Exposure response array for subacute to subchronic inhalation exposure to dichloromethane (log Y**
7004 **axis)**

Hepatic effects (hepatic vacuolation and necrosis, hemosiderosis, hepatocyte degeneration) are the critical dose-dependent noncancer effects associated with inhalation exposure to dichloromethane. These effects were seen in mice (Mennear et al., 1988; NTP, 1986) and rats (Mennear et al., 1988; Nitschke et al., 1988a; NTP, 1986; Burek et al., 1984), but not in Syrian golden hamsters (Burek et al., 1984). Inhalation bioassays with Sprague-Dawley rats identified the lowest inhalation LOAEL for nonneoplastic liver lesions in the database: 500 ppm (6 hours/day, 5 days/week for 2 years) (Nitschke et al., 1988a; Burek et al., 1984), and Nitschke et al. (1988a) identified a NOAEL of 200 ppm in female rats. Based on the results reviewed above, nonneoplastic liver lesions (specifically, hepatic vacuolation) in rats are identified as the critical noncancer effect from chronic dichloromethane inhalation in animals. Because Nitschke et al. (1988a) examined a range of exposures that included doses at the low end of the rangecompared with the range examined in Burek et al. (1984), the former study was selected as the principal study for derivation of a chronic inhalation RfC.

Reproductive performance (e.g., as assessed by number of litters, resorption rate, fetal survival, and growth) was not affected in two generations of F344 rats exposed to up to 1,500 ppm for 14 or 17 weeks before mating of the F0 and F1 generations, respectively (Nitschke et al., 1988b) or in a study of Swiss-Webster mice or Sprague-Dawley rats exposed to 1,250 ppm on GDs 6–15 (Schwetz et al., 1975). A decrease in fertility index was seen in the 150 and 200 ppm groups in a study of male Swiss-Webster mice exposed via inhalation for 6 weeks prior to mating (Raje et al., 1988), but the statistical significance of this effect varied considerably depending on the statistical test used in this analysis. Two types of developmental effects, decreased offspring weight at birth and changed behavioral habituation of the offspring to novel environments, were seen in Long-Evans rats following exposure to 4,500 ppm for 14 days prior to mating and during gestation (or during gestation alone) (Bornschein et al., 1980; Hardin and Manson, 1980). This dose was the only exposure dose used in this study. Schwetz et al. (1975) did not observe an adverse effect on gross development or soft tissue abnormalities in a study involving exposure to 1,250 ppm on GD 6 in Swiss-Webster mice or Sprague-Dawley rats, but an increase in delayed ossification of the sternebrae was seen.

Neurological impairment was not seen in lifetime rodent bioassays involving exposure to airborne dichloromethane concentrations of ≤2,000 ppm in F344 rats (Mennear et al., 1988; NTP, 1986), ≤3,500 ppm in Sprague-Dawley rats (Nitschke et al., 1988a; Burek et al., 1984), or ≤4,000 ppm in $B6C3F_1$ mice (Mennear et al., 1988; NTP, 1986). It should be noted, however, that these studies did not include standardized neurological or neurobehavioral testing. The sole subchronic or chronic study in which neurobehavioral batteries were utilized found no effects in an observational battery, a test of hind-limb grip strength, a battery of evoked potentials, or brain, spinal cord, or peripheral nerve histology in F344 rats exposed to concentrations up to 2,000 ppm for 13 weeks, with the tests performed beginning 65 hours after the last exposure (Mattsson et al., 1990).

Other effects associated with lifetime inhalation exposure to dichloromethane include renal tubular degeneration in F344 rats exposed to ≥2,000 ppm, testicular atrophy in male B6C3F$_1$ mice exposed to 4,000 ppm, and ovarian atrophy in female B6C3F$_1$ mice exposed to ≥2,000 ppm (Mennear et al., 1988; NTP, 1986). No effects on histologic, clinical chemistry, urinalysis, or hematologic variables were found in Syrian golden hamsters exposed to concentrations up to 3,500 ppm for 2 years, with the exception that the mean COHb percentage of exposed hamsters was about 30%, compared with values of about 3% in controls (Burek et al., 1984).

5.2.2. Derivation Process for Reference Concentration Values

The derivation process used for the RfC parallels the process described in section 5.1.2 on the RfD derivation; consideration of dose metrics was described in section 5.1.3. As was noted in the RfD discussion, the mechanistic issues with respect to noncancer health effects represents data gaps in the understanding of the health effects of dichloromethane.

5.2.3. Methods of Analysis—Including Models (PBTK, BMD, etc.)

The modified rat PBTK model of Andersen et al. (1991), described in Appendix C and also used in the derivation of the RfD (Figure 5-2), was used for calculating internal dosimetry of inhaled dichloromethane in Sprague-Dawley rats. Simulations of 6 hours/day, 5 days/week inhalation exposures used in the Nitschke et al. (1988a) study were performed to calculate average daily internal liver doses (Table 5-5). In the absence of data for group- and sex-specific BWs, reference values for male and female Sprague-Dawley rats in chronic studies were used (U.S. EPA, 1988a).

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Table 5-5. Incidence data for nonneoplastic liver lesions (hepatic vacuolation) and internal liver doses, based on various metrics, in female Sprague-Dawley rats exposed to dichloromethane via inhalation for 2 years (Nitschke et al., 1988a)

			Rat internal liver dose[b]			
Sex	Exposure (ppm)	Liver lesion incidence[a]	CYP	GST	GST and CYP	Parent AUC
Male	0	22/70 (31)				
	50	Not reported	Not modeled because results for middle two doses were not reported			
	200	Not reported				
	500	28/70 (40)				
Female (BW = 229 g)	0	41/70 (59%)	0	0	0	0
	50	42/70 (60%)	280.3	6.3	286.6	1.2
	200	41/70 (58%)	656.5	93.2	749.7	17.8
	500	53/70 (76%)[c]	772.6	359.0	1,131.6	68.7

[a]Number affected divided by total sample size.

[b]Internal doses were estimated using a rat PBTK model using exposures reported by study authors (50 ppm = 174 mg/m^3, 200 ppm = 695 mg/m^3, and 500 ppm = 1737 mg/m^3) and are weighted-average daily values for 1 week of exposure @ 6 h/day, 5 day/week. CYP dose is in units of mg dichloromethane metabolized via CYP pathway/L tissue/day; GST dose is in units of mg dichloromethane metabolized via GST pathway/L tissue/day.; GST and CYP dose is in units of mg dichloromethane metabolized via CYP and GST pathways/L tissue/day; and Parent AUC dose is in units of mg dichloromethane*hrs)/L tissue.

[c]Significantly ($p < 0.05$) different from control with Fisher's exact test.

Source: Nitschke et al., 1988a.

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7069 As described in section 5.1.2, the internal dose metric used was based on total hepatic
7070 metabolism through the CYP2E1 pathway (mg dichloromethane metabolized via CYP
7071 pathway/L liver/day). Figure 5-7 shows the comparison between inhalation external and internal
7072 doses, using this dose metric for the rat and the human.

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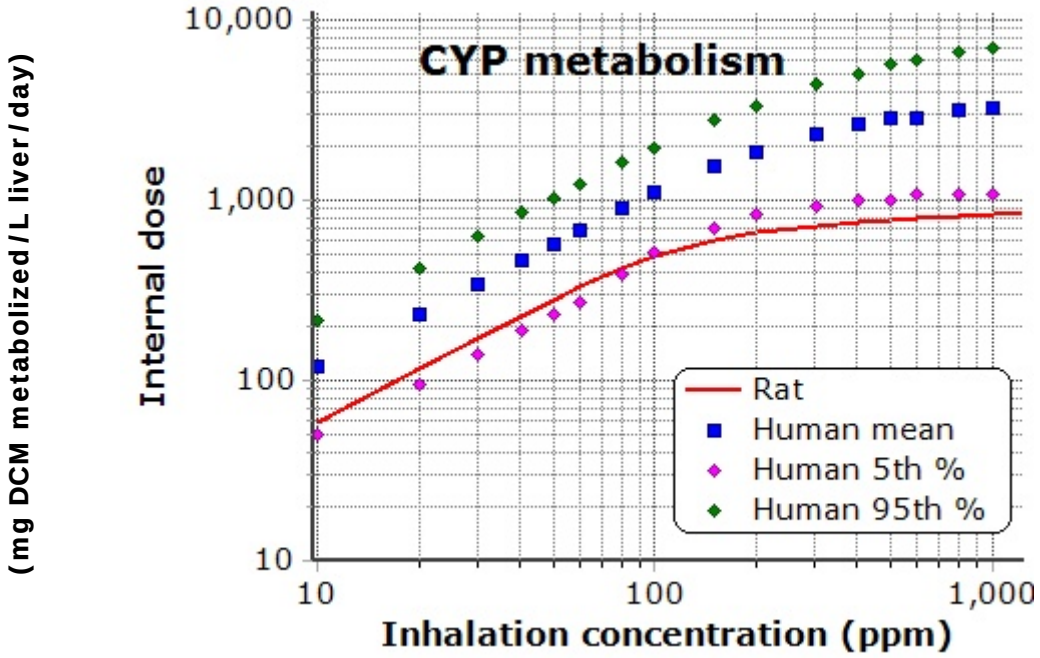

Figure 5-7. PBTK model-derived internal doses (mg dichloromethane metabolized via the CYP pathway/L liver/day) in rats and humans versus external exposures (ppm). Average daily doses were calculated from simulated rat exposures of 6 hours/day, 5 days/week, while simulated human exposures were continuous. The human metabolism rates were estimated using a computational sample of 1000 individuals per dose, including random samples of the three GST-T1 polymorphisms (+/+, +/-, -/-) in the current U.S. population based on data from Haber et al. (2002). Since a different set of samples was used for each dose, some stochasticity is evident as the human points (values) do not fall on smooth curves.

The seven dichotomous dose-response models available in EPA BMDS version 2.0 were fit to the female rat liver lesion incidence of Nitschke et al. (1988a) and PBTK model-derived internal dose data to derive rat internal BMD_{10} and the associated $BMDL_{10}$ values (Table 5-6). The quantal model is identical to the one-stage multistage model; therefore, it is not included in this set of models. A BMR of 10% was selected because, in the absence of information regarding the magnitude of change in a response that is thought to be minimally biologically significant, a BMR of 10% is generally recommended, as it provides a consistent basis of comparison across assessments. There are no additional data to suggest that the critical response has a greater sensitivity that would warrant a lower BMR. The log-probit model was the best fitting model for the female incidence data, based on AIC value among models with adequate fit. Modeling results are shown in detail in Appendix D-2).

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Table 5-6. BMD modeling results for incidence of noncancer liver lesions in female Sprague-Dawley rats exposed to dichloromethane by inhalation for 2 years, based on liver specific CYP metabolism metric (mg dichloromethane metabolized via CYP pathway per liter liver tissue per day)

Model[a]	BMD_{10}	$BMDL_{10}$	χ^2 goodness of fit *p*-value	AIC
Gamma[a]	614.27	225.96	0.48	367.22
Logistic	274.58	150.43	0.14	369.77
Log-logistic[a]	697.90	499.42	0.94	365.90
Multistage (3)[a]	506.94	153.13	0.25	368.53
Probit	275.49	152.52	0.14	369.75
Log-probit[a]	**728.96**	**523.94**	**0.98**	**365.82**
Weibull[a]	706.45	487.45	0.95	365.87

[a]These models in EPA BMDS version 2.0 were fit to the rat dose-response data shown in Table 5-5 by using internal dose metrics calculated with the rat PBTK model. Gamma and Weibull models restrict power ≥1; Log-logistic and Log-probit models restrict to slope >1, multistage model restrict betas ≥0; lowest degree polynomial with an adequate fit reported (degree of polynomial in parentheses).

Bolded model is the best-fitting model in the most sensitive sex (males), which is used in the RfC derivation.

Source: Nitschke et al., (1988a).

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7103 As with the RfD derivation, the human-equivalent internal $BMDL_{10}$ was obtained by
7104 dividing this rat internal dose metric by a pharmacokinetic scaling factor based on a ratio of BWs
7105 (scaling factor = 4.09) (Table 5-7). This scaling factor was used because the metric is a rate of
7106 metabolism rather than the concentration of putative toxic metabolites, and the clearance of these
7107 metabolites may be slower per volume tissue in the human compared with the rat. A probabilistic
7108 PBTK model for dichloromethane in humans, adapted from the model of David et al. (2006) as
7109 described in Appendix B, was then used with Monte Carlo sampling to calculate distributions of
7110 chronic HECs (in units of mg/m^3) associated with the internal $BMDL_{10}$, based on the responses
7111 in female Sprague-Dawley rats. Estimated mean, first, and fifth percentiles of this distribution
7112 are shown in Table 5-7.

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Table 5-7. Inhalation RfC for dichloromethane based on PBTK model-derived probability distributions of human inhalation exposure extrapolated from nonneoplastic liver lesion data for female rats exposed via inhalation for 2 years, based on liver-specific CYP metabolism dose metric (mg dichloromethane metabolized via CYP pathway per liter liver tissue per day)

Model[a]	Rat internal $BMDL_{10}$[b]	Human internal $BMDL_{10}$[c]	HEC (mg/m^3)[d]			Human RfC (mg/m^3)[e]
			1st percentile	5th percentile	Mean	
Log-probit	523.94	128.10	16.63	20.89	47.36	0.2

[a]Based on the best-fitting model from Table 5-6.

[b]Rat dichloromethane PBTK model-derived internal liver dose associated with lower bound on 10% extra risk for developing hepatocyte vacuolation.

[c]Human dichloromethane internal liver dose, derived by dividing the rat internal $BMDL_{10}$ by a scaling factor of 4.09 $[(BW_{human}/BW_{rat})^{0.25}]$ to account for potential interspecies pharmacokinetic differences in the clearance of metabolites.

[d]PBTK model-derived distributions of long-term, daily average airborne dichloromethane concentrations predicted by the PBTK model to yield an internal dose in humans equal to the dichloromethane internal $BMDL_{10}$.

[e]Human candidate RfC, based on female rat data, derived by dividing the 1st percentile of HEC values by a total UF of 100: 3 ($10^{0.5}$) for possible toxicodynamic differences between species, 3 ($10^{0.5}$) for variability in human toxicodynamic response, and 10 for database deficiencies. The 1st percentile point of departure is a stable estimate of the lower end of the distribution. Use of this value in the lower tail replaces use of a UF for human toxicokinetic variability.

5.2.4. RfC Derivation—Including Application of Uncertainty Factors (UFs)

The 1st percentile point of departure is a stable estimate of the lower end of the distribution. Use of this value associated with a sensitive human population addresses the uncertainty associated with human toxicokinetic variability. The RfC was calculated by dividing the first percentile of the HEC distribution in Table 5-7 by a composite UF of 100 (3 [$10^{0.5}$] to account for uncertainty about interspecies toxicodynamic equivalence, 3 [$10^{0.5}$] to account for uncertainty about toxicodynamic variability in humans, and 10 for database deficiencies). The resulting RfC was 0.2 mg/m^3 based on liver lesions in female Sprague-Dawley rats in Nitschke et al. (1988a). In deriving this RfC, factors for the following areas of uncertainty were considered:

- *Uncertainty in extrapolating from laboratory animals to humans (UF_A).* The use of PBTK models to extrapolate internal doses from rats to humans reduces toxicokinetic uncertainty in extrapolating from the rat liver lesion data but does not account for the possibility that humans may be more sensitive than rats to dichloromethane due to toxicodynamic differences. A UF of 3 ($10^{0.5}$) to account for this toxicodynamic uncertainty was applied, as shown previously in Table 5-7.

- *Uncertainty about variation in human toxicokinetics (UF_H).* The probabilistic human PBTK model used in this assessment incorporates the best available information about variability in toxicokinetic disposition of dichloromethane in humans but does not account for humans who may be sensitive due to toxicodynamic factors. Thus, a UF of 3 ($10^{0.5}$) was applied to account for possible toxicodynamic differences in sensitive humans.

Uncertainty in extrapolating from LOAELs to NOAEL (UF_L). A UF for extrapolation from a LOAEL to a NOAEL was not applied because BMD modeling was used to determine the POD, and this factor was addressed as one of the considerations in selecting the BMR. The BMR was selected based on the assumption that it represents a minimum biologically significant change.

- *Uncertainty in extrapolating from subchronic to chronic durations (UF_S).* The derived RfD is based on results from a chronic-duration drinking water toxicity study. No cross-duration UF is necessary.

- *Uncertainty reflecting incompleteness of the overall database (UF_D).* A UF of 10 was selected to address the deficiencies in the dichloromethane toxicity database. The inhalation database for dichloromethane includes several well-conducted chronic inhalation studies. In these chronic exposure studies, the liver was identified as the most sensitive noncancer target organ in rats (Nitschke et al., 1988a; NTP, 1986; Burek et al.,

1984). The critical effect of hepatocyte vacuolation was corroborated in the two principal studies (Nitschke et al., 1988a; Burek et al., 1984), which identified 500 ppm as the lowest inhalation LOAEL for noncancer liver lesions. Gross signs of neurologic impairment were not seen in lifetime rodent inhalation bioassays for dichloromethane at exposure levels up to 4,000 ppm (see section 4.2.2.2 for references), and no exposure-related effects were observed in an observational battery, a test of hind-limb grip strength, a battery of evoked potentials, or histologic examinations of nervous tissues in F344 rats exposed to dichloromethane concentrations as high as 2,000 ppm (Mattson et al., 1990). A two-generation reproductive study in F344 rats reported no effect on fertility index, litter size, neonatal survival, growth rates, or histopathologic lesions at exposures ≥100 ppm dichloromethane (Nitschke et al., 1988b). Fertility index (measured by number of unexposed females impregnated by exposed males per total number of unexposed females mated) was reduced following inhalation exposure of male mice to 150 and 200 ppm dichloromethane for 2 hours/day for 6 weeks, but the statistical significance of this effect varied considerably depending on the statistical test used in this analysis. (Raje et al., 1988). The available developmental studies include single-dose studies that use relatively high exposure concentrations (1,250 ppm in Schwetz et al. [1975]; 4,500 ppm in Hardin and Manson [1980]; and 4,500 ppm in Bornschein et al.[1980]). In one of the single-dose studies, decreased offspring weight at birth and changed behavioral habituation of the offspring to novel environments were seen following exposure of adult Long-Evans rats to 4,500 ppm for 14 days prior to mating and during gestation (or during gestation alone) (Bornschein et al., 1980; Hardin and Manson, 1980). CO, a known developmental neurotoxicant, is produced through the CYP2E1 metabolic pathway for dichloromethane. Schwetz et al. (1975) reported increased concentrations (~10% higher compared with controls) in maternal blood COHb levels in mice and rats exposed during GDs 6–15. A chronic exposure study in F344 rats reported a dose-related increase in blood COHb in females exposed to 50, 200, and 500 ppm, beginning with the first measure taken after 6 months of exposure (Nitschke et al., 1988a). The increase was seen at the lowest exposure group (50 ppm). Anders and Sunram (1982) reported elevated CO levels in maternal and fetal blood in rats following exposure to 500 ppm for 1 hour on GD 21; levels were similar in the maternal and fetal samples. Placental transfer of dichloromethane was also seen, although levels were lower in the fetus. The results from the single dose developmental toxicity study in rats (Bornschein et al., 1980; Hardin and Manson, 1980), in addition to the known increase in CO, the placental transfer of dichloromethane, and the relatively high activity of CYP2E1 in the brain compared to the liver of the developing human fetus (Hines, 2007; Brzezinski et al., 1999; Johnsrud et al., 2003), raise uncertainty regarding possible neurodevelopmental toxicity from gestational exposure to inhaled dichloromethane. In

addition, Aranyi et al. (1986) demonstrated evidence of immunosuppression following a single 100 ppm dichloromethane exposure for three hours in CD-1 mice. This study used a functional immune assay that is directly relevant to humans (i.e., increased risk of Streptococcal pneumonia-related mortality and decreased clearance of Klebsiella bacteria). No effects were seen with 50 ppm exposure for either 1 or 5 days. Systemic immunosuppression was not seen in a 4-week, 5,000 ppm inhalation exposure study, measuring the antibody response to sheep red blood cells in Sprague-Dawley rats (Warbrick et al., 2003). These studies suggest a localized, portal-of-entry effect within the lung rather than a systemic immunosuppression. Therefore, in consideration of the entire database for dichloromethane, a database UF of 10 was selected. This UF accounts for the lack of neurodevelopmental toxicity studies and developmental toxicity studies at low doses.

5.2.5. Previous RfC Assessment

No RfC was derived in the previous IRIS assessment.

5.2.6. RfC Comparison Information

A candidate RfC, based on a different approach to accounting for human toxicokinetic variability is similar to thederived RfC of 0.2 mg/m^3. Use of the mean value on the HEC distribution (47.36), with an additional UF of 3 ($10^{0.5}$) to account for human toxicokinetic variability, would yield an RfC of 0.2 mg/m^3.

For an additional comparison, an RfC was derived based on neurological endpoints from human occupational exposures. Cherry et al. (1983) compared 56 exposed and 36 unexposed workers at an acetate film manufacturing plant for dichloromethane inhalation exposure, blood levels of dichloromethane, subjective self-reporting of general health, and two objective, quantitative measurements of neurological function (digit symbol substitution and simple reaction time). The exposed and unexposed individuals were matched to within 3 years of age. The measured dichloromethane concentrations from personal breathing zone sampling of the exposed workers ranged from 28 to 173 ppm. No information on exposure duration was given, and Cherry et al. (1983) did not indicate if the exposure measurements were indicative of historical exposure levels. There were no significant differences between exposed and unexposed workers in subjective or objective measurements collected at the beginning of the work shift on a Monday (after 2 nonworking days). Exposed workers showed a slightly slower (but not significant) score than the control workers on a reaction time test, but the scores did not deteriorate during the shift. These findings suggest that repeated inhalation exposures in the range of 28–173 ppm do not result in significant effects, but the actual duration of exposure of the workers is uncertain. In the absence of data for the mean exposure levels, the exposure range midpoint of 101 ppm serves as a NOAEL for chronic neurological effects from dichloromethane

exposure. Thus, a candidate RfC of 3.5 mg/m^3 was derived by dividing the NOAEL of 351 mg/m^3 (101 ppm) by a composite UF of 100. A UF of 10 was applied to account for potentially susceptible individuals in the absence of quantitative information on the variability of neurological response to dichloromethane in the human population. A UF of 10 was applied for database deficiencies. The duration of exposures of acetate film workers (Cherry et al., 1983) was not reported, and a limited number of endpoints was evaluated. Further, definitive neurological batteries were not administered in chronic-duration animal bioassays.

Another candidate RfC was developed by using the neurological data from the study by potential long-term CNS effects in a study of retired aircraft maintenance workers (Lash et al., 1991). Retired aircraft maintenance workers, ages 55-75 years, employed in at least one of 14 targeted jobs (e.g., paint strippers) with dichloromethane exposure for 6 or more years between 1970 and 1984 (n = 25) were compared to a like group of workers without dichloromethane exposure (n = 21). From 1974 to 1986, when 155 measurements for dichloromethane exposure were made, mean breathing zone TWAs ranged from 82 to 236 ppm and averaged 225 ppm for painters and 100 ppm for mechanics; information on exposure levels prior to this time was not provided. The evaluation included several standard neurological tests, including physiological measurement of odor and color vision senses, auditory response potential, hand grip strength, measures of reaction time (simple, choice, and complex), short-term visual memory and visual retention, attention, and spatial ability. The exposed group had a higher score on verbal memory tasks (effect size approximately 0.45, $p = 0.11$) and lower score on attention tasks (effect size approximately –0.55, $p = 0.08$) and complex reaction time (effect size approximately –0.40, $p = 0.18$) compared with the control group. None of these differences were statistically significant. Given the sample size, however, the power to detect a statistically significant difference between the groups was very low (i.e., approximately 0.30 for an effect size of 0.40 using a two-tailed alpha of 0.05) (Cohen, 1987), and these results cannot be taken as evidence of no effect. An estimated exposure level from the study can be generated from the midpoint value from the exposure range (82–236 ppm; mean = 159 ppm), converted to 552 mg/m^3. If these results are viewed as a LOAEL and this estimated mean exposure level of 552 mg/m^3 was used, a composite UF of 1,000 would be applied for interspecies toxicodynamics (10), extrapolation from a LOAEL to a NOAEL (10), and database uncertainties (10), resulting in an RfC of 0.55 mg/m^3.

The value of the candidate RfC based on the data from Cherry et al. (1983), 3.5 mg/m^3, is approximately 15-fold higher, and the value of the candidate RfC based on the data from Lash et al. (1991), 0.55 mg/m^3 is approximately three times higher than thederived RfC of 0.2 mg/m^3, based on liver lesions in rats. The animal-derived RfC is preferable to the human-derived RfC because of the uncertainties about the exposure durations for the workers in the Cherry et al. (1983) study and uncertainties regarding the exposures and effect sizes in Lash et al. (1991), and because the RfC based on the rat data is more health protective.

Additional comparisons among the RfC and candidate values developed from other endpoints or data sets, using NOAEL/LOAEL methods, are shown in Table 5-8 and Figure 5-8.

Table 5-8. Potential points of departure with applied UFs and resulting candidate RfCs

Endpoint	Point of departure (mg/m^3) [a]	POD Type and Description[b]	UFs[c] Total UF	UF_A	UF_H	UF_L	UF_S	UF_D	RfC (mg/m^3)	Reference
Hepatocyte vacuolation, female rat	**523**	**BMD, 10% increase in incidence of liver lesion**	**100**	**3**	**3**	**1**	**1**	**10**	**0.2**	**Nitschke et al. (1988a)**
Renal tubular degeneration; NOAEL, male rat	620	NOAEL	1000	3	10	1	1	10	2.07	Mennear et al. (1988); NTP (1986)
Reproductive - fertility index; NOAEL, male mouse	20.7	No effect at POD, 16% decrease in fertility index seen at LOAEL dose	300	3	10	1	1	10	0.071	Raje et al. (1988)
Increased infection susceptibility (mortality risk), female mouse	15.5	NOAEL	3,000	3	10	1	10	10	0.005	Aranyi et al. (1986)
Increased IgM production, male and female rat	17,366	NOAEL	3,000	3	10	1	10	10	1.03	Warbrick et al. (2003)
Chronic CNS effects, human male	351	NOAEL	100	1	10	1	1	3	3.51	Cherry et al. (1983)
CNS changes, human male	552	LOAEL	1,000	1	10	10	1	3	0.55	Lash et al. (1991)

[a]POD = point of departure. For Nitschke et al. (1988a), this is based on BMD modeling of a 10% increase in liver lesions using internal liver dose metric (mg dichloromethane metabolism via CYP pathway per liter liver tissue per day) derived from a rat PBTK model. After an allometric scaling factor of 4.09 was applied, the human internal $BMDL_{10}$ was 128 mg/m^3. A probabilistic human PBTK model adapted from David et al. (2006) was used to generate a distribution of human equivalent concentrations from the human internal $BMDL_{10}$ and the first percentile of this distribution was used as the point of departure. For other rodent studies, the NOAEL or LOAEL concentration, in mg/m^3, was adjusted to a continuous exposure taking into account hours per day and days per week of exposure. This adjusted exposure was then converted to an HEC by multiplying the value by a dosimetric adjustment factor (DAF). Blood:air partition coefficients were 8.24 for humans, 19.8 for rats, and 23 for mice. Since the blood:air partition coefficients for both the mice and rats were greater than for humans, a DAF of 1 is recommended and was used. NOAELs or LOAELs were used as points of departure in human studies since the concentrations were already human exposures.

[b]Extra risk defined for incidence data as $(Incidence_1 - Incidence_0)/(1\text{-}Incidence_0)$, where 1 = dose at observed increased and 0 = background incidence

[c]UF_A = uncertainty in extrapolating from laboratory animals to humans, UF_H = uncertainty about variation from average humans to sensitive humans, UF_L = uncertainty about extrapolating from LOAEL to NOAEL, and UF_D = uncertainty reflecting incompleteness of the overall database. A UF extrapolating from subchronic to chronic durations (UF_S) was not used for any of these studies.

Bolded value is the basis of the RfC of 0.2 mg/m^3.

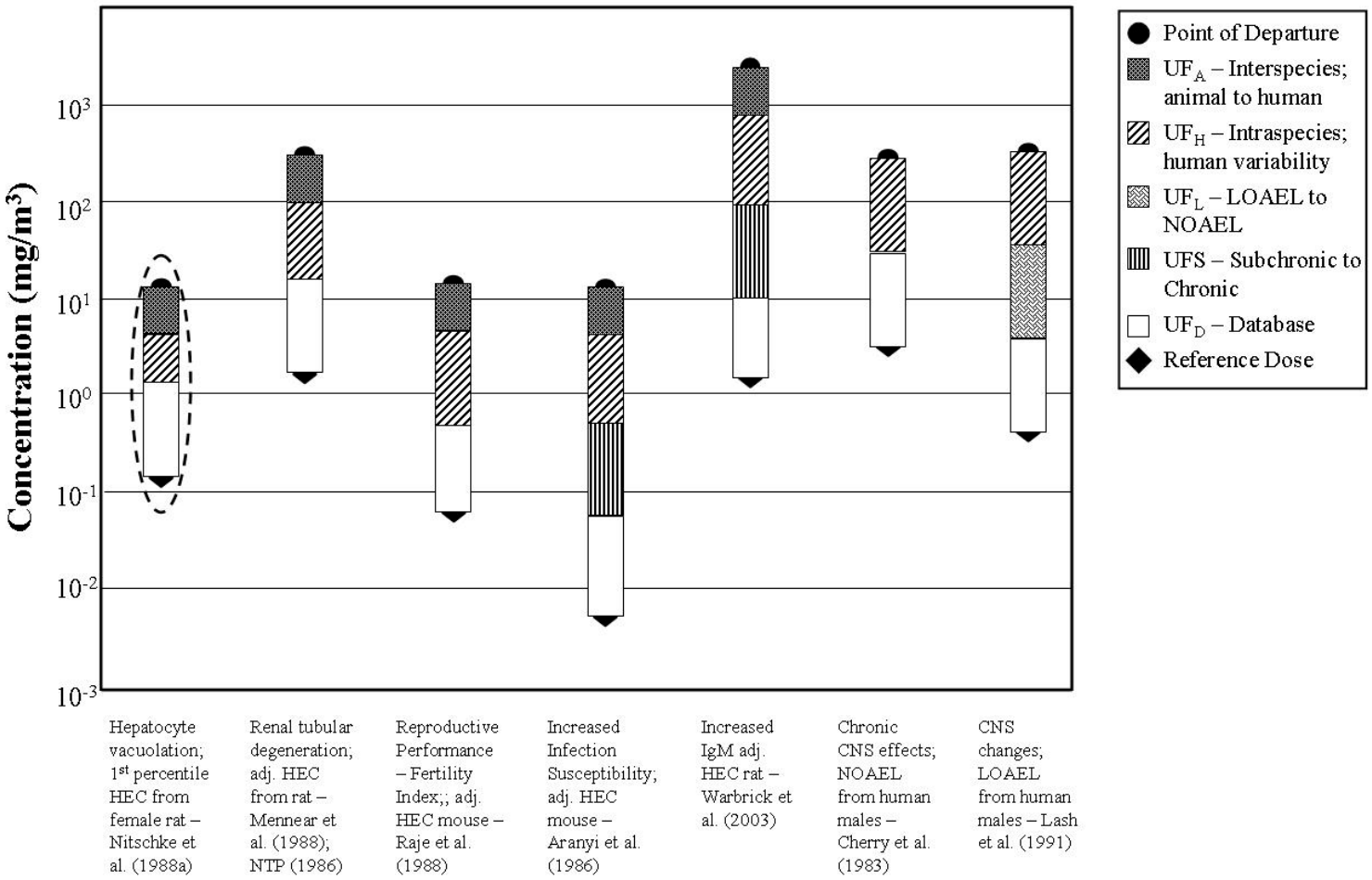

Figure 5-8. Comparison of candidate RfCs derived from selected point of departures for endpoints presented in Table 5-8.

5.3. UNCERTAINTIES IN THE ORAL REFERENCE DOSE AND INHALATION REFERENCE CONCENTRATION

Risk assessments need to include a discussion of uncertainties associated with the derived toxicity values. For dichloromethane, uncertainties related to inter- and intraspecies differences in toxicodynamics and database deficiencies are treated quantitatively via the UF approach (U.S. EPA, 1994b). Uncertainties in the toxicokinetic differences of dichloromethane between species and within humans are reduced by application of the PBTK models for rats and humans. These and other areas of uncertainty of the derived RfD and RfC are discussed below.

Adequacy of database for derivation of RfD and RfC

As summarized in sections 4.6.1.1 and 4.6.2.1, data from the available human studies on the health effects from occupational inhalation exposures provide some, but not conclusive, evidence of long-term health consequences of chronic dichloromethane exposure, specifically with respect to neurologic and hepatic damage. These data are not adequate for derivation of an RfD or RfC. However, a broad range of animal toxicology data is available for the hazard assessment of dichloromethane, as described in chapter 4. The database of oral (Table 4-35) and inhalation (Tables 4-36 and 4-37) toxicity studies includes numerous chronic, subchronic, acute, reproductive, and developmental studies. Liver toxicity in multiple rodent species is consistently identified as the most sensitive noncancer effect from oral and inhalation exposure to dichloromethane. In addition to the oral and inhalation toxicity data, there are numerous studies describing the toxicokinetics of dichloromethane. Consideration of the available dose-response data to determine an estimate of oral exposure that is likely to be without an appreciable risk of adverse noncancer health effects over a lifetime has led to the selection of noncancer liver lesions in the 2-year drinking water study in F344 rats (Serota et al., 1986a) as the critical effect and principal study for deriving the RfD for dichloromethane. The critical effect selected for the derivation of the chronic RfC is also hepatic lesions; two different studies in Sprague-Dawley rats (Nitschke et al.,1988a; Burek et al., 1984), spanning overlapping exposures, reported data on hepatic vacuolation, and the lower exposure study was chosen as the principal study (Nitschke et al.,1988a).

A critical data uncertainty was identified for neurodevelopmental effects. Animal bioassays have not identified gross or microscopic effects on neural tissues from long-term exposures or single (Schwetz et al., 1975) or multigenerational (Nitschke et al., 1988b) developmental toxicity studies. However, behavioral changes were observed in pups born to rats exposed to high levels (4,500 ppm) of dichloromethane (Bornschein et al., 1980; Hardin and Manson, 1980); lower exposures were not examined in this study. Uncertainty exists as to the development of neurological effects from lower gestational exposures in animals or humans. In addition, a critical data uncertainty has been identified that relates to potential immunotoxicity, specifically immunosuppression seen as a localized portal–of-entry effect within the lung with an acute inhalation exposure. The lack of data on immune effects from

longer-term exposure represents a significant data gap and is of particular importance because of the potential importance of immunosuppression with respect to response to infections and tumor surveillance. The weight of evidence for nonneoplastic effects in humans and animals suggests that the development of liver lesions is the most sensitive effect, with a UF applied because of the lack of neurodevelopmental studies and, for the RfC, the uncertainty regarding immunotoxicity.

Dose-response modeling

The selection of the BMD model(s) for the quantitation of the RfD and RfC does not lead to significant uncertainty in estimating the point of departure. It should be noted, however, that a level of uncertainty is inherent given the lack of data in the region of the BMR.

Interspecies extrapolation of dosimetry and risk

The extrapolation of internal dichloromethane dosimetry from nonneoplastic rat responses to human risk was accomplished using PBTK models for dichloromethane in rats and humans. Uncertainties in rat and human dosimetry used for RfD and RfC derivation can arise from uncertainties in the PBTK models to accurately simulate the toxicokinetics of dichloromethane for animals under bioassay conditions and humans experiencing relatively low, chronic environmental exposures.

There is uncertainty associated with the pharmacokinetic data used for model parameter estimation and structure validation. The data are primarily measurements of parent dichloromethane kinetics (e.g., blood or closed-chamber air concentrations over time), rather than measurements of metabolite levels which can be unambiguously attributed to one of the two principal metabolic pathways (GST and CYP). For the mouse model in particular, only parent dichloromethane data were used, though exhaled amounts of CO_2 and CO are available. Marino et al. (2006) did include data from mice pre-treated with *trans*-1,2-dichloroethylene (tDCE), a specific CYP 2E1 inhibitor, but the authors assumed without verification that *100%* of the CYP 2E1 activity was eliminated by the inhibitor when using those data. In contrast, Mathews et al. (1997) found that pretreatment of F344 rats by tDCE (100 mg/kg ip) only yielded 65% inhibition of CYP 2E1. If a significant fraction of the CYP 2E1 activity was not eliminated in the dichloromethane experiments, then that activity is erroneously assigned to the GST pathway in the parameter estimation Marino et al. (2006).

In addition to the possibility of incomplete inhibition of CYP 2E1 effecting the data interpretation, the Michaelis-Menten rate equation used in all of the published PBTK models for dichloromethane, including that of Marino et al. (2006), has in fact *not* been shown to accurately describe the CYP 2E1-mediated metabolism of dichloromethane in the relevant concentration range. While Michaelis-Menten kinetics usually describe CYP-mediated oxidation data quite well, the approach of Marion et al. (2006) implicitly assumes that any metabolism *not* described by the Michaelis-Menten equation is GST-mediated. If pathway-specific metabolite data were used to define or bound

the ratio of GST to CYP metabolism, the resulting estimates would be less sensitive to what otherwise might be small errors in the CYP rate equation. But in the modeling of the mouse data by Marino et al. (2006), the fraction of total metabolism assigned to the CYP pathway depends quite strongly on the assumed form of the CYP rate equation, along with the assumption of 100% inhibition by tDCE. In EPA's modeling of the rat in vivo PK data, using the same model structure and equations, a set of parameter values could not be found which described both the parent dichloromethane kinetics and the total amount of CO exhaled at both high and low exposure levels; in particular see panel C of Figure C-3 and note discrepancy between model and 50 mg/kg data. That the model does not describe well the dose-dependent shift in metabolism shown by those CO data suggests that the dose-dependence of the CYP Michaelis-Menten rate-equation is not adequate. As will be shown, an alternative equation for CYP kinetics may fit the existing dichloromethane data better than Michaelis-Menten kinetics, with the result that a higher portion of total dichloromethane metabolism would be interpreted as being CYP-mediated. Thus there is uncertainty in the choice of equation for the CYP pathway, which leads to uncertainty in the estimated GST:CYP metabolic ratio, upon which current risk predictions are based.

The potential error in assuming Michaelis-Menten kinetics for CYP-mediated oxidation of dichloromethane is reinforced by examining the in vitro oxidative (i.e., CYP-specific) kinetics of dichloromethane reported by Reitz et al. (1989). When extrapolated from in vitro to in vivo, the *apparent* values of the oxidative saturation constant, Km, identified by Reitz et al. (1989) for mice, rats, and humans are over 2 orders of magnitude greater than those obtained from in vivo PBTK modeling. Part of the explanation for this apparent discrepancy lies in the disparate concentration ranges investigated: Reitz et al. (1989) used much higher dichloromethane concentrations in vitro than those observed in or predicted for the various in vivo pharmacokinetic studies. In particular, the oxidation of dichloromethane could involve *two* oxidative processes, one with a high affinity (low Km) corresponding to the nonlinearity observed in vivo and one with a low affinity (high Km) corresponding to the nonlinearity observed in vitro. Further, the low-affinity process would have nearly linear kinetics in the exposure range used for the in vivo dosimetry studies and hence be difficult to distinguish from GST-mediated metabolism unless pathway-specific metabolite data are used. One can hypothesize that this second oxidative process is not inhibited by tDCE and hence corresponds to the 35% of oxidative metabolism which was observed to remain in rats after tDCE treatment by Mathews et al. (1997).

The data of Reitz et al (1989) could simply indicate a second CYP with low-affinity dichloromethane activity. However that possibility is contradicted by the results of Kim and Kim (1996) who observed that another CYP 2E1-specific inhibitor, disulfiram, completely abolished dichloromethane-induced increases on COHb in rats. Another possible explanation which would support the findings observed in Kim and Kim (1996) as well as Reitz et al (1989) and the various in vivo data is that a number of CYPs exhibit "atypical" kinetics, not described by the classic Michaelis-Menten equation, consistent with the enzymes having dual binding sites as proposed by Korzekwa et al

(1988). (Korzekwa et al. (1988) demonstrated atypical kinetics for several CYP-isozyme/substrate pairs, but not specifically for CYP 2E1.) Figure 5-9 shows kinetic model fits to the in vitro mouse dichloromethane oxidation kinetic data of Reitz et al. (1989), after expressing those data on a per gram of liver basis. Both the standard Michaelis-Menten kinetic equation (solid line) and the dual-binding equation (dashed line) given by Korzekwa et al. (1988) are shown. In particular, the high-affinity (low) Km for the dual-binding equation was set equal to that obtained by Marino et al. (2006) from their PBTK modeling. This figure shows that the dual binding model is not only consistent with the *apparent* high-affinity saturation obtained from in vivo PBTK modeling (Km of Marino et al. (2006)), but also with the apparent low-affinity (high Km) data of Reitz et al. (1989), and describes those in vitro data better than the standard Michaelis-Menten equation. (Reitz et al. (1989) used classic Lineweaver-Burk plots to display their kinetic data; i.e., 1/reaction rate vs. 1/concentration. The systematic discrepancy between their data and Michaelis-Menten kinetics evident in Figure 5-9 is much less obvious with that scaling, which likely explains why they made no note of it.)

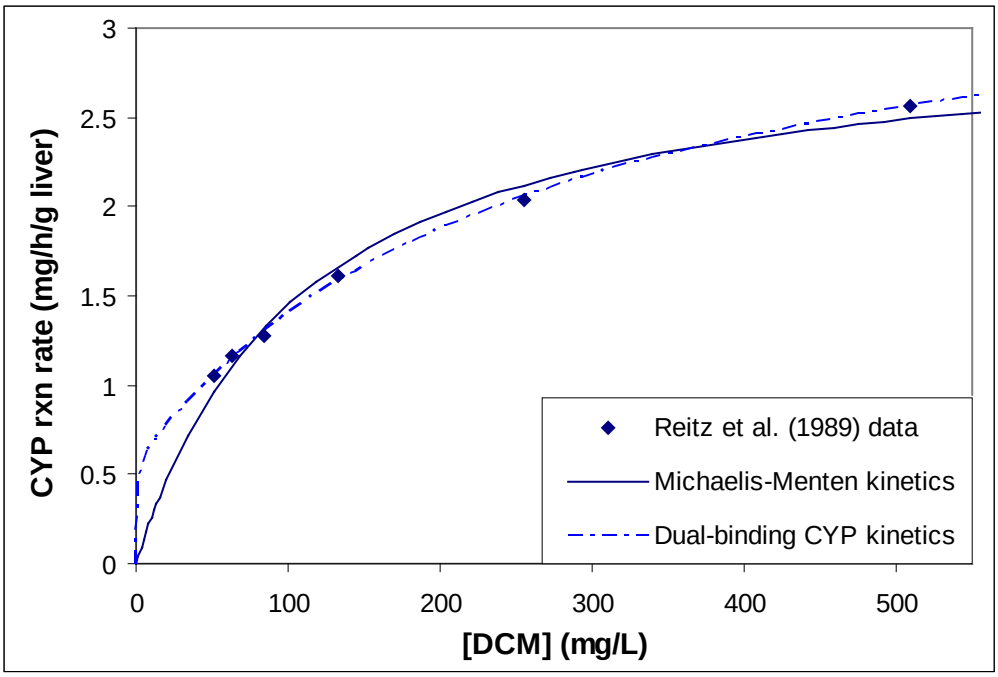

Figure 5-9. Comparison of dichloromethane oxidation rate data with alternate kinetic models. Dichloromethane (DCM) oxidation data obtained with mouse liver microsomes by Reitz et al. (1989) (points), expressed on a per gram of liver basis, are shown with a fitted Michaelis-Menten equation (solid line) or a fitted dual-binding-site equation as described by Korzekwa et al. (1988) (dashed line), where the high affinity saturation constant of the dual-binding-site equation set equal to the mean Km determined for mice via PBTK modeling by Marino et al. (2006). The Km for the Michaelis-Menten equation (108 mg/L) is inconsistent with the in vivo

DCM dosimetry data, while the in vitro data shown here are inconsistent with the Km estimated in vivo (0.42 mg/L) if that equation is used.

In summary regarding model equations, the current PBTK model used the standard Michaelis-Menten equation to describe CYP 2E1-catalyzed oxidation of small volatile organic compounds. Analysis of the dichloromethane (pharmaco)kinetic data and evaluation of the inconsistencies describe above suggest that an alternate equation, which would impact risk predictions, may better represent CYP 2E1-induced oxidation of dichloromethane. However, this hypothesis requires further laboratory testing, for example, by measuring dichloromethane oxidation in a bacterial expression system where only CYP 2E1 is expressed over a concentration range sufficient to firmly distinguish between the two kinetic forms indicated in the figure above. Until such experiments are conducted, the existing PBTK model remains the best available science for dose- and hence risk-extrapolation from rodents to humans. Still, this model structure uncertainty implies uncertainty in the quantitative results obtained with the model. Analysis of the GST-mediated metabolism of dichloromethane measured by Reitz et al. (1988) shows that those results are within a factor of three of the GST kinetic parameters used in the current PBTK model, indicating that the any error in the GST:CYP balance is no greater than that, a reasonable level of uncertainty.

One other component of quantitative uncertainty arises in examining the results of the Bayesian modeling for the human PBTK model of David et al. (2006). The authors reported Bayesian posterior statistics for the population average of each fitted parameter when calibration was performed either with specific published data sets or the entire combined data set. While one would generally expect that the values obtained from the combined data set should be a weighted average of the values from individual data sets, the population mean for the liver GST activity (coefficient), K_{FC}, was 0.852 while the values from the individual data sets ranged from 1.92-34.0 $kg^{0.3}/h$.

A clarification provided by D. Marino (personal communication)[8] is that the parameter bounds stated in the text of David et al. (2006) were only applied for the analysis of the DiVincenzo and Kaplan (1981) and the combined data set. But according to the text and distribution prior statistics specified, the upper bound for K_{FC} would have been 12 $kg^{0.3}/h$ (mean + 2.5 standard deviations (SDs), with mean = 2 and SD = mean*CV = 2*2 = 4). The data of Andersen et al. (1991) were not used in the combined analysis because only group average values were available from that source, rather than individual data. Since the remaining study-specific mean K_{FC} values were 7.95, 5.87, 34.0, and 1.92, with CVs of less than 2, it seems unlikely that application of this upper bound would result in a value of K_{FC} of only 0.852 $kg^{0.3}/h$. Given that there had been convergence problems with the combined data set when parameter values were unbounded, it is possible that convergence had not actually been reached after

[8] Email from Dale Marino to Glinda Cooper dated April 25, 2007.

parameter bounds were introduced, and a higher value for K_{FC} would have been obtained had the chain been continued longer.

Since the numerical average of the mean K_{FC} values for the four data sets included in the combined data set was 12.4 and the upper bound was 12, the impact of using an intermediate value of K_{FC}, specifically the DiVincenzo and Kaplan value of 5.87 $kg^{0.3}/h$ was explored. Changing only the K_{FC} is not realistic since the dichloromethane data effectively define total metabolism (sum of CYP and GST pathways) and there is naturally a negative correlation between the predicted CYP metabolic rate and the GST metabolic rate, required to describe this total. Therefore, it would be inconsistent with the dichloromethane data to increase K_{FC} without adjusting the CYP metabolic rate downward, and likewise all other parameters. The distributions for all of the fitted parameters were rescaled by the ratio of the mean for DiVincenzo and Kaplan (1981) to the mean for the combined data set (e.g., the distribution for K_{FC} was multiplied by 5.87/0.852, the ratio of the two posterior means). The resulting predicted upper-bound (95^{th} and 99^{th} percentile) GST metabolism rates for a fixed level of exposure (1 $\mu g/m^3$ inhaled concentration or 1 mg/kg/day oral exposure) in the GST-T1 +/+ population increased by more than a factor of 10. (For inhalation exposure the mean value also increased by over 10-fold, but for oral exposure the mean increased by only 2-fold.) Since the majority of metabolism occurs via the CYP pathway at these low levels, there is not a proportionate (i.e., over 10-fold) decrease in that rate, but HEC and HED calculations increased by 10-30% for the mixed GST-T1 population, depending on the route of exposure and distribution statistic compared. Thus the impact of this model uncertainty appears to be relatively small for the noncancer assessment, but quite large for the cancer assessment.

The dose metric used in the models is the rate of metabolism to a putative toxic metabolite, rather than the concentration average or area under the concentration curve of the metabolite, so the model specifically fails to account for rodent-human differences in clearance or removal of the toxic metabolite. A scaling factor based on BW ratios, was used to account for this difference.

The rat model was modified and utilized in a deterministic manner. Data were not available to perform a hierarchical Bayesian calibration in the rat. Thus, uncertainties in the rat model predictions had to be assessed qualitatively. To address these uncertainties, a sensitivity analysis was conducted to determine which model parameters most influence the predictions for a given dose metric and exposure scenario.

Sensitivity is a measure of the degree to which a given model output variable (i.e., dose metric) is influenced by perturbation in the value of model parameters. The approach implemented was a univariate analysis in which the value of an individual model parameter was perturbed by an amount (Δ), in the forward and reverse direction (i.e., an increase and decrease from the nominal value), and the change in the output variable was determined. Sensitivity coefficients were calculated as follows:

$$f'(x) \approx \frac{f(x+\Delta x) - f(x)}{\Delta x} \cdot \frac{x}{f(x)} \qquad \textbf{(Eq. 5-1)}$$

where *x* is the model parameter *f(x)* is the output variable, Δx is the perturbation of the parameter from the nominal value, and *f'(x)* is the sensitivity coefficient. In equation 5-1, the sensitivity coefficients are scaled to the nominal value of *x* and *f(x)* to eliminate the potential effect of units of expression. Therefore, the sensitivity coefficient is a measure of the proportional (unitless) change in the output variable produced by proportional change in the parameter value. Parameters that have higher sensitivity coefficients have greater influence on the output variable. They are considered more sensitive than parameters with lower values. The results of the sensitivity analysis are useful for assessing uncertainty in model predictions, based on the level of confidence or uncertainty in the model parameter(s) to which the dose metric is most sensitive.

Sensitivity coefficients for the noncancer dose metric (mg dichloromethane metabolized via CYP-mediated pathway/L liver/day), were determined for each of the model parameters; a similar analysis was also done for a metric based on the GST-mediated pathway. Sensitivity analyses for both oral and inhalation exposures were performed. The exposure conditions were set to be near or just below the lowest bioassay exposure resulting in significant increases in the critical effect.

For the CYP-mediated metabolism from oral exposure, the VLC and VSC (liver volume and slowly perfused tissue volume, respectively) parameters exert the largest influence (Figure 5-10). The high influence of these two parameters was due to the fact that the dose metric is a tissue-specific rate of metabolism, the majority of CYP metabolism is attributed to the liver, and that changes in liver volume have a greater impact on the total CYP metabolism that the individual Vmax value. For inhalation exposures VMAXC, in addition to VLC and VSC have the highest sensitivity coefficients (Figure 5-11). The physiological parameters (VLC and VSC) are known with a high degree of confidence (Brown et al., 1997). V_{MAXC} for the rat was estimated by fitting to the PK data as described in Chapter 3 and Appendix C, subject to model structure/equation uncertainties as detailed above, and hence is known with less certainty than the physiological parameters. That total exhaled CO, which is a specific to the CYP pathway, is within 50% of measured levels (Fig. C-8, panel C), however, provides a similar level of confidence in the balance between CYP and GST pathways predicted by the rat PBTK model.

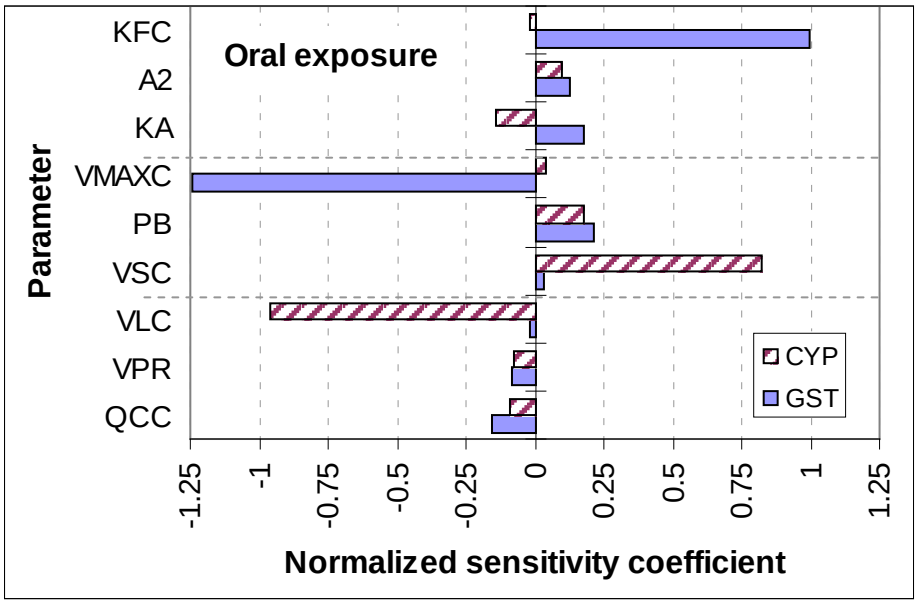

Figure 5-10. Sensitivity coefficients for long-term mass CYP- and GST-mediated metabolites per liver volume from a daily drinking water concentration of 10 mg/L in rats. KFC = GST-mediated metabolism rate; A2 = proportion of liver GST metabolism attributed to the lung; KA = oral absorption rate from gut; VMAXC = CYP-mediated maximum rate of metabolism; PB = blood:air partition coefficient; VSC = slowly perfused tissue volume; VLC = liver volume; VPR = Ventilation perfusion ratio ; QCC = cardiac output constant.

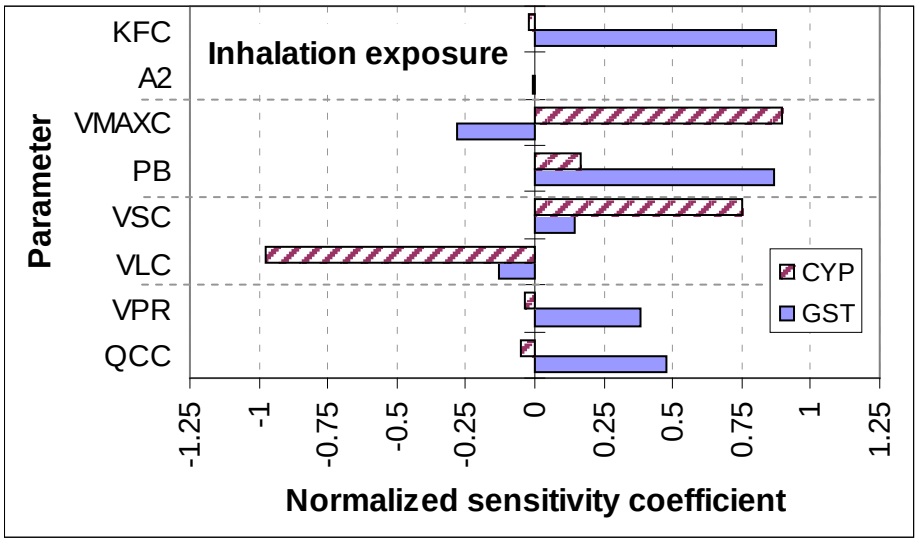

Figure 5-11. Sensitivity coefficients for long-term mass CYP- and GST-mediated metabolites per liver volume from a long-term average daily inhalation concentration of 500 ppm in rats. (KA is not included since it has no impact on inhalation dosimetry.) KFC = GST-mediated metabolism rate; A2 = proportion of liver GST metabolism attributed to the lung; VMAXC = CYP-mediated maximum rate of metabolism; PB = blood:air partition coefficient; VSC = slowly perfused tissue volume; VLC = liver volume; VPR = Ventilation perfusion ratio ; QCC = cardiac output constant.

In summary, the uncertainties associated with use of the rat PBTK model should not markedly affect the values of the RfD and RfC based on the metrics considered. An additional uncertainty results from the lack of knowledge concerning the most relevant dose metric (e.g., a specific metabolite) for the non-cancer endpoints considered. This basic research question represents a data gap. This uncertainty was addressed by considering different dose metrics (CYP metabolism alone, GST metabolism alone, sum of GST and CYP, and the AUC of the parent compound). The GST metabolism and the AUC dose metrics did not present reasonable choices based on model fit and consistency of response across studies at comparable dose levels. Given these results, the combination of hepatic metabolism through the GST and the CYP pathways would not be expected to result in an improvement to a metric based only on CYP metabolism. The CYP-metabolism dose metric seems to be most consistent with the data., and so is the metric chosen for the RfD and RfC derivations.

Sensitive human populations

The potential for sensitivity to dichloromethane in a portion of the human population due to pharmacokinetic differences was addressed quantitatively by using a human probabilistic PBTK model, as modified by the EPA, to generate distributions of human exposures likely to result in a specified internal $BMDL_{10}$. The model and resulting distributions take into account the known non-chemical-specific variability in human physiology as well as total variability and uncertainty in dichloromethane-specific metabolic capability. The first percentile values of the distributions of human equivalent doses (Table 5-3) and HECs (Tables 5-7) served as points of departure for candidate RfDs and RfCs, respectively, to protect toxicokinetically sensitive individuals. Selection of the first percentile allows generation of a stable estimate for the lower end of the distribution. The mean value of the human equivalent oral dose in Table 5-3 was about two-fold higher than the corresponding first percentile values, and the mean value of human equivalent inhalation concentration in Table 5-7 was approximately three-fold higher than the first percentile value. The internal dose metric in the analyses described in these tables was the mg dichloromethane metabolized via the CYP pathway per liter liver per day, and thus the comparisons of the first percentile and mean values give estimates of the amount of variability in the population to metabolize dichloromethane by the CYP metabolic pathways on a liver-specific basis. The mean: 1^{st} percentile ratios for these distributions is attributed to the dependence of the dose metric on hepatic blood flow rate (metabolism being flow limited). This blood flow is expected to be highly and tightly correlated with liver volume, resulting in very similar delivery of dichloromethane per volume liver across the population. While the mean: 1^{st} percentile ratios for the oral distribution is less than the default intra-human toxicokinetic UF of 3, it is quite similar to that obtained by Sweeney et al. (2003) for acrylonitrile, where an extensive sensitivity analysis indicated a 99^{th} percentile:mean ratio of less than 2.2 among several internal dose metrics. The population-structured distributions for physiological parameters and broadened distributions for metabolic

parameters used here provide a good degree of confidence that the population variability has not been under-estimated.

The internal dose metric used in the RfD and RfC derivations was based on the rate of CYP metabolism. GST-T1 polymorphisms could affect this rate, as the GST-T1 null genotype would be expected to result in an increase in the metabolism through the CYP pathway, resulting in a greater sensitivity to a CYP-related effect. The effect of GST variability on the RfD and RfC values was examined by comparing results obtained specifically for the GST-T1 null genotype to those obtained for the population of mixed genotypes. The values for human equivalent doses and HECs were very similar for these two groups (e.g., mean HEC 47.36 and 47.49 for the mixed and the GST-$T1^{-/-}$ null genotypes, respectively; 1^{st} percentile HEC 16.63 and 16.69 for the mixed and the GST-$T1^{-/-}$ null genotypes, respectively), and use of this population would not result in a change in the recommended RfD or RfC.

As a further level of sensitivity analysis, we compared model predictions of the human equivalent dose, as listed in Table 5-3, for the general population (estimates covered 0.5- to 80-year-old male and female individuals) to three subpopulations: 1-year-old children (males and females), 70-year-old men, and 70-year-old women. For the general population and each subpopulation a Monte Carlo simulation representing 10,000 individuals was conducted, and histograms of the resulting distribution of human equivalent administered doses are shown in Figure 5-12, with corresponding statistics in Table 5-9. All groups used in these comparisons were limited to the GST-$T1^{-/-}$.

The results shown above for differences in human equivalent dose values in different populations are qualitatively what would be expected: a relatively broad distribution for the general population with specific populations representing narrower components of that distribution. There are some differences between men and women at 70 years of age, but neither of these would be greatly misrepresented by the general population estimate. While 1-year-old children represent more of a distinct tail in the general population, in this case the distribution of human equivalent concentrations in the general estimate is lower than that seen in what would otherwise be considered a more sensitive population. This difference most likely results from the higher specific respiration rate in children versus adults, which allows them to eliminate more of orally ingested dichloromethane by exhalation, leading to lower internal metabolized doses

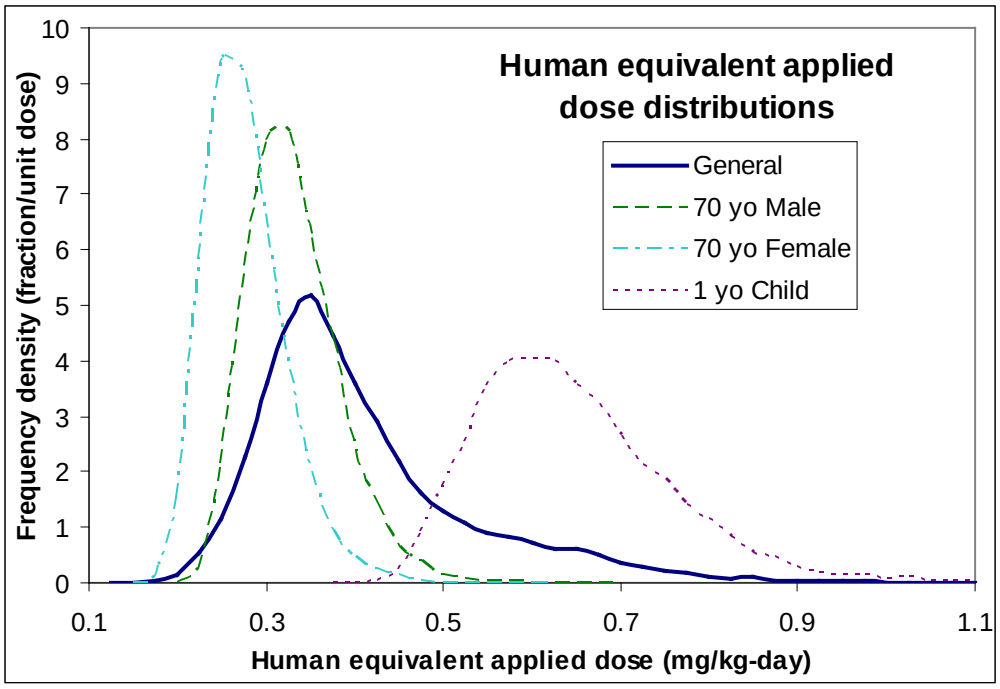

Figure 5-12. Frequency density of human equivalent applied doses in specific populations in comparison to a general population (0.5- to 80-year-old males and females) estimate for an internal dose of 15.1 mg dichloromethane metabolized by CYP per liter liver per day; all groups were restricted to the GST-T1 $^{-/-}$ population).

Table 5-9. Statistical characteristics of human equivalent applied doses in specific populations of the GST-T1 $^{-/-}$ group

	Human equivalent applied dose (mg/kg-day)[a]		
Population	**Mean**	**5th percentile**	**1st percentile**
All ages[b]	3.95×10^{-1}	2.52×10^{-1}	2.14×10^{-1}
1-year-old children	6.34×10^{-1}	4.87×10^{-1}	4.54×10^{-1}
70-year-old men	3.18×10^{-1}	2.48×10^{-1}	2.29×10^{-1}
70-year-old women	2.64×10^{-1}	2.03×10^{-1}	1.85×10^{-1}

[a]Exposure levels predicted to result in 15.1 mg dichloromethane metabolized via CYP pathway per liter liver per day (based on mean $BMDL_{10}$ across acceptable models from Table 5-3).
[b]0.5- to 80-year-old males and females.

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A similar comparison was made for inhalation HEC values, as shown in Figure 5-13 and Table 5-10. For HEC values, the distributions for 70-year-old men and women are both virtually indistinguishable from the general population, and while 1-year-old children are clearly distinct, they are

less different than in the human equivalent administered dose comparison and in this case are more sensitive than the population in general. As described in detail in Appendix B, the allometric alveolar ventilation constant, QAlvC, is about 28 L/hour-kg$^{0.75}$ in a 1-year-old child but averages around 14 L/hour-kg$^{0.75}$ in an adult. Combining this with the difference between a BW of 10 kg in that child and 70 kg in an "average" adult, the respiration rate per kg BW is about threefold higher in the child versus adult. As noted above, for oral exposures this leads to faster elimination by respiration in children, while for inhalation exposures it leads to higher uptake for a given air concentration.

The lack of difference in elderly adults versus the general population in HEC values is likely due to the fact that the rate of exposure and rates of metabolism (the latter being the key dose metric) both scale as BW$^{0.75}$, with the scaling coefficients being either similar (respiration) or identical (metabolism) among adults, who comprise the majority of the population, while for oral exposures the exposure rate is normalized to total BW and scales as BW^{1}, while elimination routes increase as BW$^{0.75}$. Moreover, oral exposures are simulated as occurring in a series of bolus exposures (drinking episodes) during the day, and the higher body-fat content occurring in the elderly (see Appendix B) means that such a dose that might saturate metabolism and therefore have a higher fraction exhaled in a leaner individual will tend to be more sequestered in fat and slowly released, resulting in a higher fraction metabolized (less saturation of metabolism) in a more obese individual. The difference among adults of different ages for dosimetry from oral ingestion (bolus exposure) will be greater than the difference for inhalation exposures. More careful examination of Figure 5-13 shows that the distribution for 70-year-old women, for whom the fat fraction is estimated to be greatest, has a lower peak and higher upper tail than for the general population. So the physiological differences do have some impact that is qualitatively consistent with what is seen from oral exposure, given the mechanistic considerations described here. But the impact of those differences is far less for inhalation exposure.

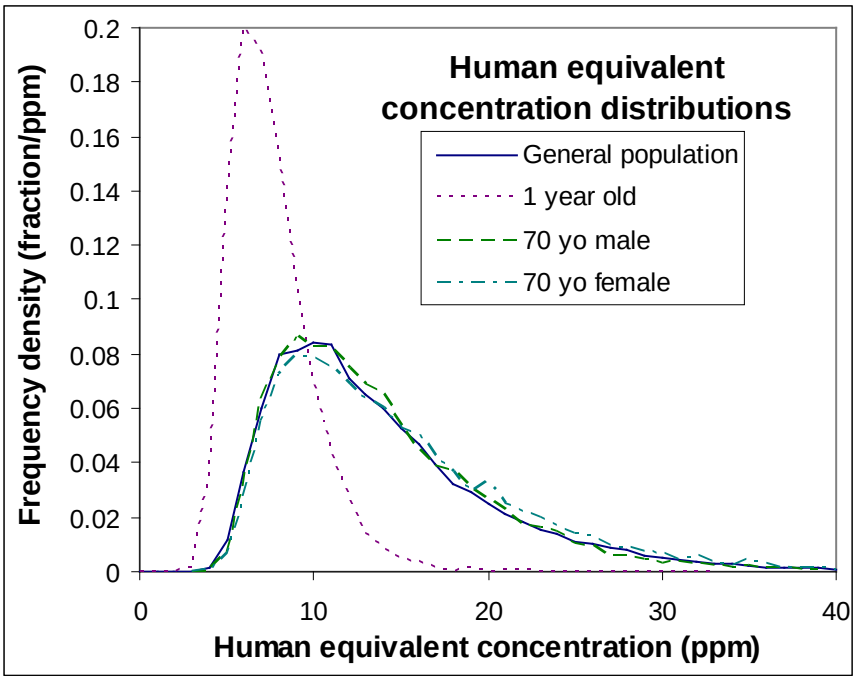

Figure 5-13. Frequency density of HECs in specific populations in comparison to a general population (0.5- to 80-year-old males and females) estimate for an internal dose of 128.1 mg dichloromethane metabolized by CYP per liter liver per day ; all groups restricted to the GST-T1 $^{-/-}$ population).

Table 5-10. Statistical characteristics of HECs in specific populations of the GST-T1 $^{-/-}$ group

	HEC (mg/m^3)[a]		
Population	**Mean**	**5th percentile**	**1st percentile**
All ages[b]	47.4	20.9	16.6
1-year-old children	24.7	14.4	12.1
70-year-old men	46.4	21.7	17.8
70-year-old women	50.0	22.2	18.0

[a]Exposure levels predicted to result in 128.1 mg dichloromethane metabolized via CYP pathway per liter liver per day (based on mean $BMDL_{10}$ across acceptable models from Table 5-7).
[b]0.5- to 80-year-old males and females.

No data are available regarding toxicodynamic differences within a human population. Therefore, a UF of 3 for possible differences in human toxicodynamic responses is intended to be protective for sensitive individuals.

5.4. CANCER ASSESSMENT

5.4.1. Cancer Oral Slope Factor

5.4.1.1. *Choice of Study/Data—with Rationale and Justification*

No human data are available for the quantification of potential neoplastic effects from oral exposures to dichloromethane. In the only chronic (2-year) oral exposure cancer bioassay, significant increases in the incidence of liver adenomas and carcinomas was observed in male, but not female, $B6C3F_1$ mice exposed by drinking water, with incidence rates of 19, 26, 30, 31, and 28% in groups with estimated mean intakes of 0, 61, 124, 177, and 234 mg/kg-day, respectively (trend p- value = 0.058) (see Table 4-38 for group comparisons) (Serota et al., 1986b; Hazelton Laboratories, 1983). Evidence of a trend for increased risk of liver tumors (described as neoplastic nodule or hepatocellular carcinoma) was seen in female F344 rats, but not males, exposed via drinking water ($p < 0.01$) (Serota et al., 1986a). However, the potential malignant characterization of the nodules was not described, and no trend was seen in the data limited to hepatocellular carcinomas. The derivation of the cancer oral slope factor (OSF) is based on the male mouse data (Serota et al., 1986b; Hazelton Laboratories, 1983) because of their greater sensitivity compared to female mice and to male and female rats. The study authors concluded that these increases were "within the normal fluctuation of this type of tumor incidence". However, the trend *p*-value for these results is of borderline statistical significance and it may not be reasonable to apply a correction for multiple comparisons given the lack of independence of the groups and given a specific focus on the liver as a target organ. The development of liver tumors in $B6C3F_1$ mice is associated with metabolite production in this tissue via the GST metabolic pathway (section 4.7.3), a pathway that also exists in humans. Modeling intake, metabolism, and elimination of dichloromethane in mice and humans is feasible. Thus, it is reasonable to apply the best available PBTK models to estimate equivalent internal doses in mice and humans.

5.4.1.2. *Derivation of Oral Slope Factor*

In a manner similar to the derivation of the noncancer toxicity values, PBTK models for dichloromethane in mice and humans were used in the derivation of toxicity values (cancer OSF and IUR) for cancer endpoints based on lung (for inhalation) and liver (for oral and inhalation) tumor data in the mouse (Figure 5-14). A deterministic PBTK model for dichloromethane in mice was first used to convert mouse drinking water or inhalation exposures to long-term daily average values of internal lung-specific GST metabolism (GST metabolism in lung/lung volume) or liver-specific GST metabolism (GST metabolism in liver/liver volume). The choice of this dose metric was made based on data pertaining to the mechanism(s) involved in the carcinogenic response, specifically data supporting the involvement of a GST metabolite(s). The evidence pertaining to the GST pathway is discussed in section 4.7, and includes the enhanced genotoxicity seen in bacterial and mammalian in vitro assays with the introduction of GST metabolic capacity (Graves et al., 1994a) and the suppression by pretreatment with

a GSH depletory of the production of DNA SSBs seen in acute inhalation exposure to dichloromethane in mice (Graves et al., 1995).

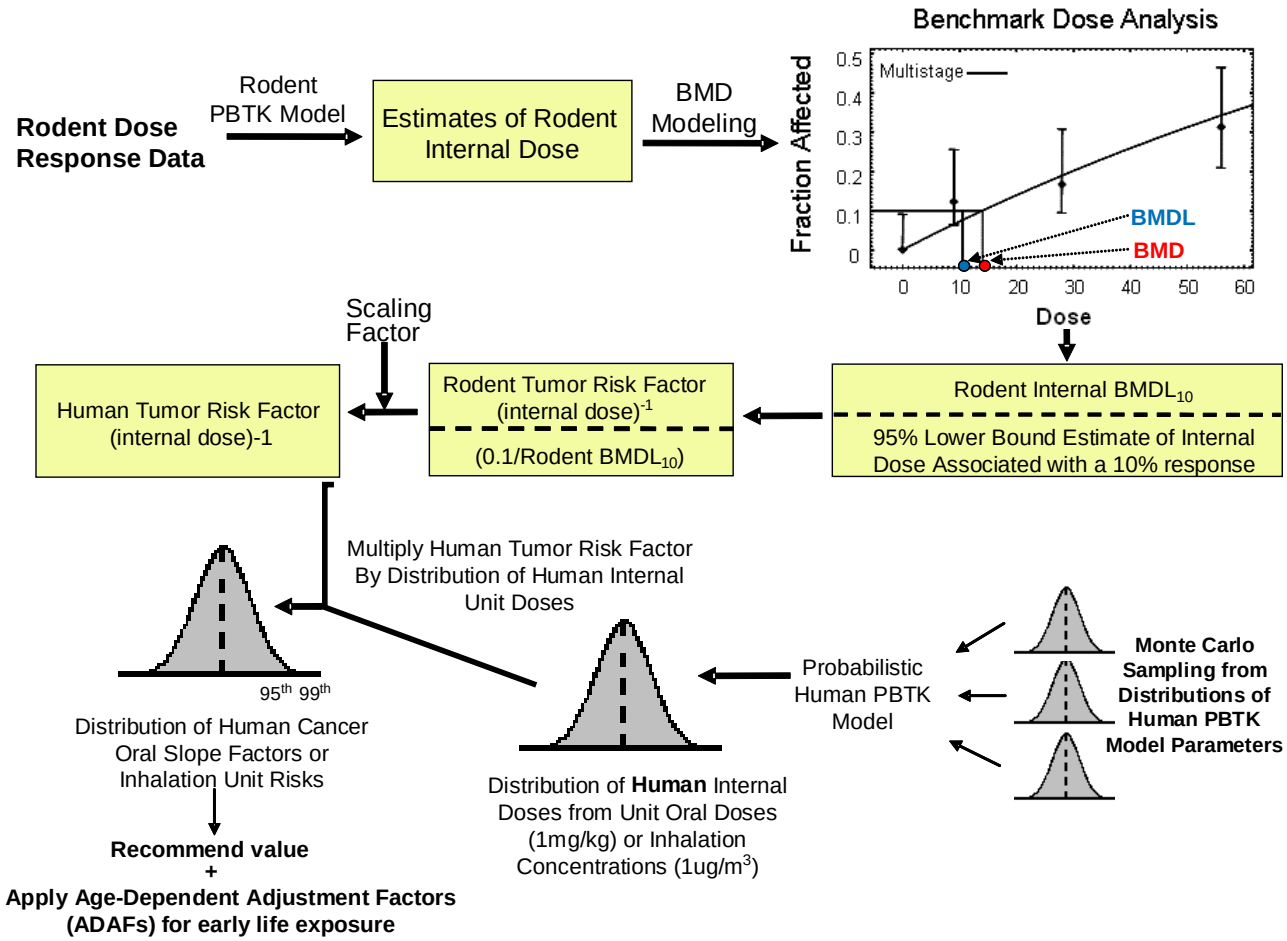

Figure 5-14. Process for deriving cancer OSFs and IURs by using rodent and human PBTK models.

The multistage cancer model (using BMDS version 2.0) was then fit to the tumor incidence data and internal dose data for rodents, and BMD_{10} and associated $BMDL_{10}$ values (for a BMR of 10% extra risk) were calculated. A probabilistic PBTK model for dichloromethane in humans, adapted from David et al. (2006) (see Appendix B), was used with Monte Carlo sampling to calculate distributions of internal lung or liver doses associated with chronic unit oral (1 mg/kg-day) or inhalation (1 $\mu g/m^3$) exposures. The resulting distribution of human internal doses was multiplied by a human internal dose tumor risk factor (in units of reciprocal internal dose) to generate a distribution of OSFs or IURs associated with a chronic unit oral or inhalation exposure, respectively.

The parameter statistics reported by David et al. (2006) include both the inter-individual variability that would have been elucidated by the Bayesian analysis (variation between mean values for each individual for which data were available) and uncertainty in those values. Since EPA's objective is to account for both population variability and parameter uncertainty, however, these statistics were primarily as-is (exceptions discussed in Appendix B) to define population distributions. Assuming that these parameters are distributed independently, ignoring the covariance that was likely represented in the actual posterior chains, will tend to over-estimate the overall range of parameters and hence distribution of dose metrics in the population, compared to what one would obtain if the covariance were explicitly included. Thus if the covariance (i.e., the variance-covariance matrix) for the set of parameters had been reported by David et al., it could have been used to narrow the predicted distribution of internal doses, or equivalent applied doses. Lacking such information the approach used will not under-estimate risk or over-estimate lower bounds on human equivalent exposure levels.

5.4.1.3. *Dose-Response Data*

Data for liver tumors in male $B6C3F_1$ mice following exposure to dichloromethane in drinking water were used to develop oral cancer slope factors (Serota et al., 1986b; Hazelton Laboratories, 1983). Significant increases in incidence of liver adenomas and carcinomas were observed in male, but not female, $B6C3F_1$ mice exposed for 2 years (Table 5-11). No significant decreases in survival were observed in the treated groups of either sex compared with controls. The at-risk study populations (represented by the denominators in the incidence data) were determined by excluding all animals dying prior to 52 weeks.

5.4.1.4. *Dose Conversion and Extrapolation Methods: Cancer Oral Slope Factor*

Dose conversion

The mouse PBTK model of Marino et al. (2006) was based on the PBTK model for dichloromethane by Andersen et al. (1987), which was modified to include dichloromethane metabolism in the lung compartment and kinetics of CO and COHb (Andersen et al., 1991). For the mouse, physiological parameters and partition coefficients were adjusted to match those

reported in Andersen et al. (1991, 1987) and Clewell et al. (1993), respectively, while QCC, VPR, and metabolic parameter distribution mean values were derived via MCMC model calibration reported by Marino et al. (2006) (Appendix B). The model of Marino et al. (2006) was used to simulate daily drinking water exposures comprising six discrete drinking water episodes for specified times and percentage of total daily intake (Reitz et al., 1997) and to calculate average lifetime daily internal doses for the male mouse data shown in Table 5-11. A first-order oral uptake rate constant (k_a) of 5 hours^{-1} was taken from Reitz et al. (1997) to describe the uptake of dichloromethane from the gastrointestinal tract to the liver. Study-specific BWs were not available, so reference BWs for male B6C3F$_1$ mice in chronic studies (U.S. EPA, 1988a) were used. Based on evidence that metabolites of dichloromethane produced via the GST pathway are primarily responsible for dichloromethane carcinogenicity in mouse liver (summarized in section 4.7.3) and the assumption that these metabolites are sufficiently reactive that they do not have substantial distribution outside the liver, the recommended selected internal dose metric for liver tumors was daily mass of dichloromethane metabolized via the GST pathway per unit volume of liver (Table 5-11). Figure 5-15 shows the comparison between internal and external doses in the liver in mice and humans. The whole-body metabolism metric was also examined. This metric would be more relevant under a scenario of slowly cleared metabolites that undergo general circulation.

Table 5-11. Incidence data for liver tumors and internal liver doses, based on GST metabolism dose metrics, in male B6C3F$_1$ mice exposed to dichloromethane in drinking water for 2 years

Sex	Nominal (actual) daily intake (mg/kg-day)	Mouse liver tumor incidence[a]	Mouse internal liver metabolism dose[b]	Mouse whole body metabolism dose[c]
Male	0 (0)	24/125 (19%)	0	0
(BW =	60 (61)	51/199 (26%)	17.5	0.73
37.3 g)	125 (124)	30/99 (30%)[d]	63.3	2.65
	185 (177)	31/98 (32%)[d]	112.0	4.68
	250 (234)	35/123 (28%)[d]	169.5	7.1

[a]Hepatocellular carcinoma or adenoma, combined. Mice dying prior to 52 weeks were excluded from the denominators. Cochran-Armitage trend p-value = 0.058.
[b]mg dichloromethane metabolized via GST pathway/L liver/day. Internal doses were estimated from simulations of actual daily doses reported by the study authors.
[c]Based on the sum of dichloromethane metabolized via the GST pathway in the lung plus the liver, normalized to total BW (i.e., [lung GST metabolism (mg/day) + liver GST metabolism (mg/day)]/kg BW. Units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day.
[d]Significantly ($p \leq 0.05$) different from control incidence by Fisher's exact test performed by Syracuse Research Corporation.

Sources: Serota et al., 1986b; Hazelton Laboratories, 1983

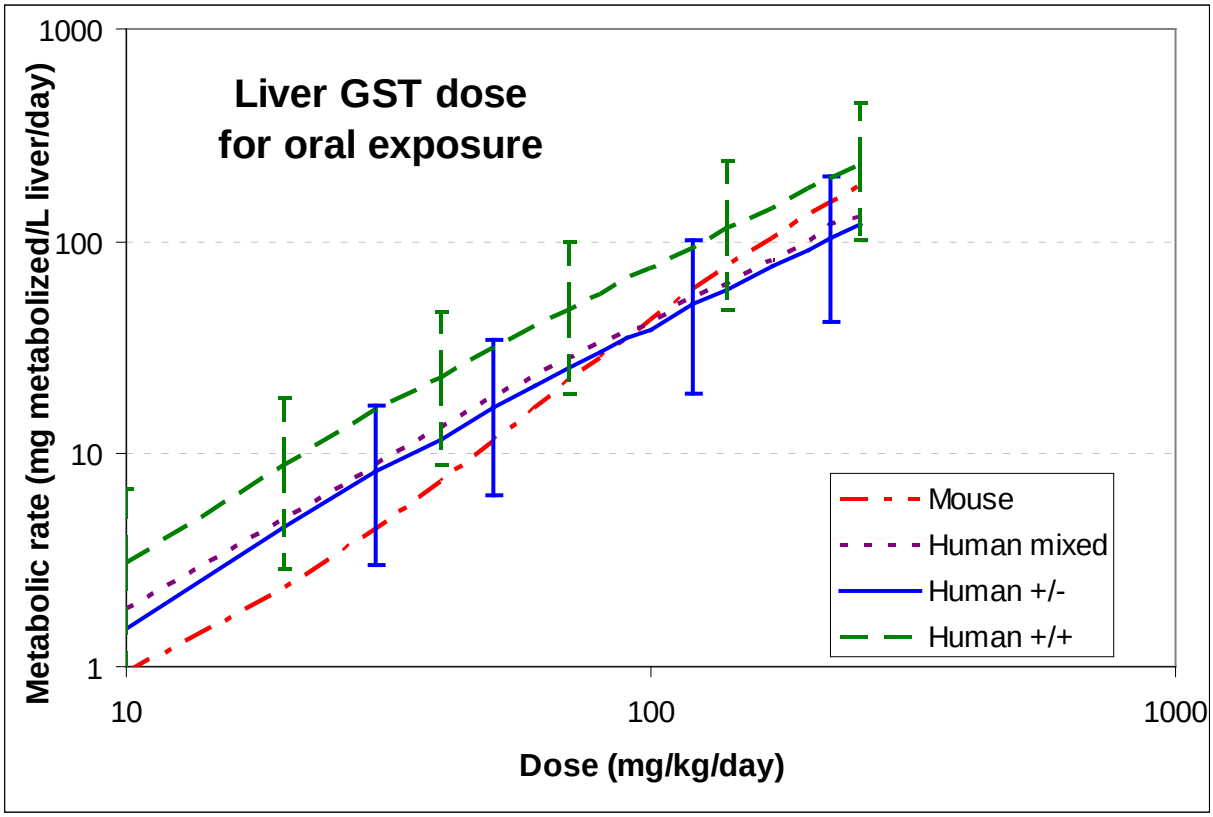

Figure 5-15. PBTK model-derived internal doses (mg dichloromethane metabolized via the GST pathway per liter liver per day) in mice and humans and their associated external exposures (mg/kg-day) used for the derivation of cancer OSFs based on liver tumors in mice. Six simulated drinking water episodes are described by Reitz et al. (1997). The human metabolism rates were estimated using a computational sample of 1000 individuals per dose, including random samples of the three GST-T1 polymorphisms (+/+, +/-, -/-; "Human mixed" curve) or samples restricted to the GST +/+ or +/- populations, in the current U.S. population based on data from Haber et al. (2002). Since a different set of samples was used for each dose, some stochasticity is evident as the human points (values) do not fall on smooth curves. Error bars indicate the range of 5th-95th percentile for the sub-populations sampled at select concentrations.

Dose-response modeling and extrapolation

The multistage dose-response model was fit to the mouse liver tumor incidence and PBTK model-derived internal dose data to derive mouse internal BMD_{10} and $BMDL_{10}$ associated with 10% extra risk (Table 5-12). Different polynomial models, and models dropping dose groups starting with the highest dose group were compared based on adequacy of model fit as assessed by overall χ^2 goodness of fit (p-value > 0.10) and examination of residuals, particularly in the region of the benchmark response (BMR). Appendix E-1 provides details of the BMD modeling results. The mouse liver tumor risk factor (extra risk per unit internal dose) was calculated by dividing 0.1 by the mouse $BMDL_{10}$ for liver tumors.

Table 5-12. BMD modeling results and tumor risk factors for internal dose metric associated with 10% extra risk for liver tumors in male B6C3F$_1$ mice exposed to dichloromethane in drinking water for 2 years, based on liver-specific GST metabolism and whole body GST metabolism dose metrics

Internal dose metric	BMDS model[b]	χ^2 goodness of fit *p*- value	Mouse BMD_{10}[c]	Mouse $BMDL_{10}$[c]	Allometric-scaled human $BMDL_{10}$[d]	Tumor Risk Factor[e]	
						Scaling = 1.0	Allometric-scaled
Liver-specific	MS (1,1)	0.56	73.0	39.6	5.66	2.53×10^{-3}	1.77×10^{-2}
Whole-body	MS (1,1)	0.56	3.05	1.65	0.24	--	4.24×10^{-1}

[a]Liver specific dose units = mg dichloromethane metabolized via GST pathway per liter tissue per day; Whole-body dose units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day)

[b]The multistage (MS) model in EPA BMDS version 2.0 was fit to the mouse dose-response data shown in Table 5-11 using internal dose metrics calculated with the mouse PBTK model. Numbers in parentheses indicate (1) the number of dose groups dropped in order to obtain an adequate fit and (2) the degree polynomial of the model.

[c]BMD_{10} and $BMDL_{10}$ refer to the BMD-model-predicted mouse internal and its 95% lower confidence limit, associated with a 10% extra risk for the incidence of tumors.

[d]Mouse $BMDL_{10}$ divided by $(BW_{human}/BW_{mouse})^{0.25} = 7$.

[e]Dichloromethane tumor risk factor (extra risk per unit internal dose) derived by dividing the BMR (0.1) by the mouse $BMDL_{10}$ and by the allometric-scaled human $BMDL_{10}$,, for the scaling = 1.0 and allometric-scaled risk factors, respectively.

7754
7755
7756 Linear extrapolation from the internal human $BMDL_{10}$ values ($0.1/BMDL_{10}$) was used to
7757 derive oral risk factors for liver tumors, based on tumor responses in male mice. Proposed key
7758 events for dichloromethane carcinogenesis are discussed in sections 4.7 and 5.4.1.1. The linear
7759 low-dose extrapolation approach for agents with a mutagenic mode of action was selected.
7760 *Application of allometric scaling factor*
7761 As discussed in section 4.7 and summarized in 5.4.1.2, several lines of evidence point to
7762 the involvement of the GST metabolic pathway in the carcinogenic response seen in
7763 dichloromethane. The role of specific metabolites has not been firmly established, however.
7764 S-(chloromethyl)-glutathione is an intermediate to the production of formaldehyde through this
7765 pathway (Hashmi et al., 1994). Formation of free hydrogen ion is also hypothesized, although
7766 no direct evidence supporting this has been presented. The pattern of HPRT gene mutations seen
7767 in CHO cells incubated with GST-complete mouse liver cytosol preparations suggest that
7768 S-(chloromethyl)glutathione, rather than formaldehyde, is responsible for the mutagenic effects
7769 associated with dichloromethane (Graves et al., 1996). DNA reaction products (e.g., DNA
7770 adducts) produced by S-(chloromethyl)glutathione have not been quantified, possibly due to
7771 potential instability of these compounds (Guengerich et al., 2003; Hashmi et al., 1994).
7772 The question of the role of specific metabolites, and particularly how these metabolites
7773 are transformed or removed is a key question affecting the choice of a scaling factor to be used in
7774 conjunction with the internal dose metric based on rate of GST metabolism. If the key metabolite
7775 is established and is known to be sufficiently reactive to not spread in systemic circulation, then

it can be assumed that 1) the level of reactivity and rate of clearance (i.e., disappearance due to local reactivity) for this metabolite, per volume tissue, is equal in rodents and humans; and (2) risk is proportional to the long-term daily average concentration of the metabolite. Under these assumptions, rodent internal $BMDL_{10}$ values based on tissue-specific dichloromethane metabolism require no allometric scaling to account for toxicodynamic differences and predict the corresponding level of human risk as a function of the metric (i.e., the scaling factor in Figure 5-14 was equal to 1.0). (A single metabolite is referenced, but the same argument holds in general for more than one metabolite). Under this scenario and assumptions, humans and rodents with the same long-term daily average metabolite formation per volume tissue (e.g., equal internal $BMDL_{10}$) should both experience the same long-term average concentration of the metabolite when the metabolite is highly reactive and hence experience the same extra risk.

Although the evidence points to a specific metabolic pathway and to site-specific actions resulting from a reactive metabolite that does not escape the tissue in which it is formed, some assumptions remain concerning this hypothesis. Specifically, the active metabolite(s) have not been established, and data pertaining to the reactivity or clearance rate of these metabolite(s) are lacking. Quantitative measurements of adducts of interest or of the half life of relevant compounds in humans and in mice are not available. To address the uncertainties in the available data it may be appropriate to use a scaling factor that addresses the possibility that the rate of clearance for the metabolite is limited by processes that are known to scale allometrically, such as blood perfusion or enzyme activity. This case would result in use of a mouse:human dose-rate scaling factor of $(BW_{human}/BW_{mouse})^{0.25} = 7$ to adjust the mouse-based $BMDL_{10}$ values downward. Using this internal dose metric (liver-specific metabolism with allometric scaling), equivalent rodent and human internal $BMDL_{10}$ values result in a human liver tumor risk factor ($0.1/BMDL_{10}$) that is assumed equal to that for the mouse, given a 70-year lifetime exposure.

Another alternative that can be used is based on an allometrically-scaled whole-body metabolism metric. In this case, less weight is given to the evidence of site-specificity, as this metric allows for systemic circulation of the relevant metabolites.

The cancer toxicity values derived using each of these metrics and scaling factors (i.e., liver-specific metabolism with and without allometric-scaling and the whole-body metabolism metric) are presented in the following tables. Considering the lack of data pertaining to clearance rates or the actual AUC of the active carcinogenic metabolite(s) in mice and humans, the OSF recommended by the EPA is based on the allometrically-scaled tissue-specific GST metabolism rate dose metric.

Calculation of OSFs

The human PBTK model adapted from David et al. (2006) (see Appendix B), using Monte Carlo sampling techniques, was used to calculate distributions of human internal dose metrics of daily mass of dichloromethane metabolized via the liver-specific GST pathway per

unit volume of liver resulting from a long-term average daily drinking water dose of 1 mg/kg dichloromethane. In another analysis of whole body metabolism, a dose metric based on the total metabolites formed in liver and lungs via GST metabolism per BW was used. The human model used parameter values derived from Monte Carlo sampling of probability distributions for each parameter, including MCMC-derived distributions for the metabolic parameters (David et al., 2006). The drinking water exposures comprised six discrete drinking-water episodes for specified times and percentage of total daily intake (Reitz et al., 1997) (Appendix B).

The distribution of cancer OSFs shown in Table 5-13 were derived by multiplying the human oral liver tumor risk factors by the respective distributions of human average daily internal doses resulting from chronic, unit oral exposures of 1 mg/kg-day dichloromethane. Because adjustments for interindividual variability are not generally used or recommended in cancer risk analysis, the mean slope factor was selected as the recommended value to be used in deterministic risk assessments; other values at the upper end of the distribution are also presented.

Consideration of Sensitive Human Subpopulations

An important issue in the derivation process used by EPA, pertaining to the use of the human PBTK model, stems from the assumption regarding the population for which the derivation should be applied. The inclusion of the GST-T1 null subpopulation in effect dilutes the risk that would be experienced by those who carry a GST-T1 allele, by averaging in non-responders (i.e., the GST-T1$^{-/-}$ genotype). Thus, the cancer OSF was derived specifically for carriers of the GST-T1 homozygous positive (+/+) genotype, that is the population that would be expected to be most sensitive to the carcinogenic effects of dichloromethane given the GST-related dose metric under consideration. In addition, cancer values derived for a population reflecting the estimated frequency of GST-T1 genotypes in the current U.S. population (20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$, the "mixed" population) are also presented. All simulations also included a distribution of CYP activity based on data from Lipscomb et al. (2003).

Table 5-13. Cancer OSFs for dichloromethane based on PBTK model-derived internal liver doses in B6C3F1 mice exposed via drinking water for 2 years, based on liver-specific GST metabolism and whole body metabolism dose metrics, by population genotype

Internal dose metric and scaling factor[a]	Population genotype[b]	Human tumor risk factor[c]	Distribution of human internal dichloromethane doses from 1 mg/kg-day exposure[d]			Resulting candidate human OSF[e] $(mg/kg\text{-}day)^{-1}$		
			Mean	95th percentile	99th percentile	Mean	95th percentile	99th percentile
Liver-specific, allometric-scaled	GST-T1$^{+/+}$	1.77×10^{-2}	0.80×10^{-1}	1.91×10^{-1}	2.89×10^{-1}	1.4×10^{-3}	3.4×10^{-3}	5.1×10^{-3}
	Mixed	1.77×10^{-2}	0.45×10^{-1}	1.39×10^{-1}	2.24×10^{-1}	8.0×10^{-4}	2.5×10^{-3}	4.0×10^{-3}
Liver-specific, scaling = 1.0	GST-T1$^{+/+}$	2.53×10^{-3}	0.80×10^{-1}	1.91×10^{-1}	2.89×10^{-1}	2.0×10^{-4}	4.8×10^{-4}	7.3×10^{-4}
	Mixed	2.53×10^{-3}	0.45×10^{-1}	1.39×10^{-1}	2.24×10^{-1}	1.2×10^{-4}	3.5×10^{-4}	5.7×10^{-4}
Whole-body , allometric-scaled	GST-T1$^{+/+}$	4.24×10^{-1}	1.90×10^{-3}	4.60×10^{-3}	7.20×10^{-3}	8.1×10^{-4}	2.0×10^{-3}	3.1×10^{-3}
	Mixed	4.24×10^{-1}	1.08×10^{-3}	3.40×10^{-3}	5.49×10^{-3}	4.6×10^{-4}	1.4×10^{-3}	2.3×10^{-3}

[a]Liver specific dose units = mg dichloromethane metabolized via GST pathway per liter tissue per day; Whole-body dose units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day.
[b]GST-T1$^{+/+}$ = homozygous, full enzyme activity; mixed = population reflecting estimated frequency of genotypes in current U.S. population: 20% GST-T$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).
[c]Dichloromethane tumor risk factor (extra risk per unit internal dose per day) derived by dividing the BMR (0.1) by the allometric-scaled human $BMDL_{10}$ and the mouse $BMDL_{10}$ for the allometric-scaled and scaling = 1.0 risk factors, respectively (from Table 5-12).
[d]Mean, 95th, and 99th percentile of the human PBTK model-derived probability distribution of daily average internal dichloromethane dose resulting from chronic oral exposure of 1 mg/kg-day.
[e]Derived by multiplying the dichloromethane tumor risk factor by the PBTK model-derived probabilistic internal doses from daily exposure to 1 mg/kg-day.

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5.4.1.5. *Oral Cancer Slope Factor*

The recommended cancer OSF for dichloromethane is 1×10^{-3} $(mg/kg\text{-}day)^{-1}$ (rounded from 1.4×10^{-3}), and is based on liver tumor responses in male $B6C3F_1$ mice exposed to dichloromethane in drinking water for 2 years (Serota et al., 1986b; Hazelton Laboratories, 1983). The OSF was derived by using a tissue-specific GST metabolism dose metric, with allometric scaling, for the population that is presumed to have the greatest sensitivity (the GST-$T1^{+/+}$ genotype). The application of ADAFs to the cancer OSF is recommended and is described in section 5.4.4.

5.4.1.6. *Alternative Derivation Based on Route-to-Route Extrapolation*

For comparison, alternative cancer OSFs were derived via route-to-route extrapolations from the data for liver tumors in male and female $B6C3F_1$ mice exposed by inhalation for 2 years (Mennear et al., 1988; NTP, 1986). This derivation, shown in Table 5-14, uses the cancer IUR derived in section 5.4.2.4 (see Table 5-19 for these IUR values) and the distribution of human internal dichloromethane exposures from 1 mg/kg-day exposure using the tissue-specific GST metabolism dose metric (mg dichloromethane metabolized via the GST pathway per liter liver per day). The oral cancer slope factor values based on the route-to-route extrapolations from liver tumors in mice exposed by inhalation (Table 5-14) are about one order of magnitude lower than those based on the liver tumor responses in mice exposed via drinking water.

Table 5-14. Alternative route-to-route cancer OSFs for dichloromethane extrapolated from male B6C3F$_1$ mouse inhalation liver tumor incidence data using a tissue-specific GST metabolism dose metric, by population genotype

Internal dose metric and scaling factor	Population Genotype[a]	Human tumor risk factor[b]	Distribution of human internal dichloromethane doses from 1 mg/kg-day exposure[c]			Resulting candidate human OSF[d] (mg/kg-day)$^{-1}$		
			Mean	95th percentile	99th percentile	Mean	95th percentile	99th percentile
Liver-specific, allometric-scaled	GST-T1$^{+/+}$	1.29×10^{-3}	0.80×10^{-1}	1.91×10^{-1}	2.89×10^{-1}	1.0×10^{-4}	2.5×10^{-4}	3.7×10^{-4}
	Mixed	1.29×10^{-3}	0.45×10^{-1}	1.39×10^{-1}	2.24×10^{-1}	5.8×10^{-5}	1.8×10^{-4}	2.9×10^{-4}
Liver-specific, scaling = 1.0	GST-T1$^{+/+}$	1.84×10^{-4}	0.80×10^{-1}	1.91×10^{-1}	2.89×10^{-1}	1.5×10^{-5}	3.5×10^{-5}	5.3×10^{-5}
	Mixed	1.84×10^{-4}	0.45×10^{-1}	1.39×10^{-1}	2.24×10^{-1}	8.3×10^{-6}	2.6×10^{-5}	4.1×10^{-5}
Whole-body metabolism	GST-T1$^{+/+}$	3.03×10^{-2}	1.90×10^{-3}	4.60×10^{-3}	7.20×10^{-3}	5.8×10^{-5}	1.4×10^{-4}	2.2×10^{-4}
	Mixed	3.03×10^{-2}	1.08×10^{-3}	3.40×10^{-3}	5.49×10^{-3}	3.3×10^{-5}	1.0×10^{-4}	1.7×10^{-4}

[a]GST-T1$^{+/+}$ = homozygous, full enzyme activity; mixed = population reflecting estimated frequency of genotypes in current U.S. population: 20% GST-T$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).

[b]Dichloromethane tumor risk factor (extra risk per milligram dichloromethane metabolized via GST pathway per liter tissue per day) derived by dividing the BMR (0.1) by the allometric-scaled human $BMDL_{10}$ and the mouse $BMDL_{10}$ for the allometric-scaled and scaling = 1.0 risk factors, respectively (from inhalation unit risk data, Table 5-19).

[c]Mean, 95th, and 99th percentile of the human PBTK model-derived probability distribution of daily average internal dichloromethane dose (milligrams dichloromethane metabolized via GST pathway per liter tissue per day) resulting from chronic oral exposure of 1 mg/kg-day.

[d]Derived by multiplying the dichloromethane tumor risk factor by the PBTK model-derived probabilistic internal doses from daily exposure to 1 mg/kg-day.

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5.4.1.7. *Alternative Based On Administered Dose*

One comparison that can be made is with an alternative OSF based on liver tumors in mice, using the external concentrations of dichloromethane in the mouse as converted to human equivalent doses and then applying this by using BMD modeling to obtain the $BMDL_{10}$ and resulting oral cancer risk. Mouse bioassay exposures were adjusted to human equivalent doses as follows:

human equivalent dose =

(nominal daily intake ÷ BW scaling factor) × daily exposure adjustment factor

where BW scaling factor = $(BW_{human}/BW_{mouse})^{0.25} = 7$

and

daily exposure adjustment factor = 5/7

The human equivalent administered doses for the 0, 60, 125, 185, and 250 mg/kg-day dose groups used in the liver tumor analysis (Table 5-11) (from Serota et al. [1986b]) were 0, 6.12, 12.75, 18.87,and 25.51 mg/kg-day, respectively. The BMD modeling and OSF derived from these values are shown in Table 5-15. The resulting OSF, based on the liver tumors in the mouse, is approximately one order of magnitude higher than the current recommended value obtained by using the mouse and human PBTK models.

Table 5-15. Cancer OSF based on a human $BMDL_{10}$ using administered dose for liver tumors in male $B6C3F_1$ mice exposed to dichloromethane in drinking water for 2 years

Sex, tumor type	BMDS model[a]	χ^2 goodness of fit *p*- value	Human BMD_{10}[c]	Human $BMDL_{10}$[c]	Cancer OSF[d] $(mg/kg\text{-}day)^{-1}$
Male, liver	MS (0,1)	0.55	19.4	10.4	1.0×10^{-2}

[a]The multistage (MS) model in EPA BMDS version 2.0 was fit to the mouse liver tumor data shown in Table 5-11. The human equivalent doses for the 0, 60, 125, 185, and 250 mg/kg-day dose groups used in the liver tumor analysis were 0, 6.12, 12.75, 18.87, and 25.51 mg/kg-day, respectively, based on application of BW scaling factor $= (BW_{human}/BW_{mouse})^{0.25} = 7$ and adjusting for daily exposure by multiplying by 5/7 days. Numbers in parentheses indicate (1) the number of dose groups dropped in order to obtain an adequate fit, starting with the highest dose group, and (2) the degree polynomial of the model.
[c]BMD_{10} and $BMDL_{10}$ refer to the BMD-model-predicted human equivalent administered dose (mg/kg-day) and its 95% lower confidence limit, associated with a 10% extra risk for the incidence of tumors.
[d]Cancer OSF (risk per mg/kg-day) = 0.1/human $BMDL_{10}$.

The administered dose methodology can be considered equivalent to using a single-compartment, whole-body model of dichloromethane, where the internal dose metric is the AUC of dichloromethane itself and clearance of dichloromethane scales from mice to humans as $BW^{0.75}$. The estimates based on the PBTK model, in contrast, use the rate of metabolism of

dichloromethane (GST) as the metric. Another difference is that the administered dose methodology does not account in any way for the GST polymorphism and so might be considered as representing the general/mixed-GST-genotype population rather than the +/+ subpopulation.

5.4.1.8. *Previous IRIS Assessment: Cancer Oral Slope Factor*

The previous IRIS assessment derived a cancer OSF of 7.5×10^{-3} $(mg/kg\text{-}day)^{-1}$ by the application of the multistage model to combined incidence of hepatocellular adenomas, carcinomas, and neoplastic nodules from two studies. These were the 2-year drinking water study of dichloromethane in $B6C3F_1$ mice by the Hazelton Laboratories (1983) and the 2-year inhalation study of dichloromethane in $B6C3F_1$ mice by NTP (1986). The slope factor was the arithmetic mean of two candidate slope factors, 1.2×10^{-2} $(mg/kg\text{-}day)^{-1}$ (Hazelton Laboratories, 1983) and 2.6×10^{-3} $(mg/kg\text{-}day)^{-1}$ (NTP, 1986). Since the NTP (1986) animal data were from inhalation exposures, the estimated inhaled doses were calculated for mice and humans (assuming near complete uptake into lung tissues and blood) and converted to administered doses in units of mg/kg-day. Assumed inhalation rates of 0.0407 and 20 m^3/day were used for mice and humans, respectively. No adjustments were made for species differences in metabolism or toxicokinetics.

5.4.1.9. *Comparison of Cancer Oral Slope Factors Using Different Methodologies*

Cancer OSFs derived using different dose metrics and assumptions are summarized in Table 5-16. The recommended OSF of 1×10^{-3} per mg/kg-day (rounded to one significant digit) is based on a tissue-specific GST-internal dose metric, with allometric scaling (= 7), because of some uncertainty regarding the rate of clearance of the relevant metabolite(s) formed via the GST pathway. The value derived specifically for the GST-$T1^{+/+}$ population is recommended to provide protection for the population that is hypothesized to be most sensitive to the carcinogenic effect. The values based on the GST-$T1^{+/+}$ group are approximately two-fold higher than those for the full population for the dose metrics used in this assessment (Table 5-16). Within a genotype population, the values of the OSF among most of the various dose metrics vary by about one to two orders of magnitude.

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Table 5-16. Comparison of OSFs derived using various assumptions and metrics, based on tumors in male mice

Population[a]	Dose metric	Species, sex	Tumor	Scaling factor	Mean OSF $(mg/kg\text{-}day)^{-1}$	Source (Table)
GST-T1$^{+/+}$	**Tissue-specific GST-metabolism rate[b]**	**Mouse, male**	**Liver**	**7.0**	**1.4×10^{-3}**	**Table 5-13**
	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	2.0×10^{-4}	Table 5-13
	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	8.1×10^{-4}	Table 5-13
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	7.0	1.0×10^{-4}	Table 5-14
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	1.0	1.5×10^{-5}	Table 5-14
	Route-to-route extrapolation, whole-body metabolism	Mouse, male	Liver	7.0	5.8×10^{-5}	Table 5-14
Mixed	Tissue-specific GST-metabolism rate[b]	Mouse, male	Liver	7.0	8.0×10^{-4}	Table 5-13
	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	1.2×10^{-4}	Table 5-13
	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	4.6×10^{-4}	Table 5-13
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	7.0	5.8×10^{-5}	Table 5-14
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	1.0	8.3×10^{-6}	Table 5-14
	Route-to-route extrapolation, whole-body metabolism	Mouse, male	Liver	7.0	3.3×10^{-5}	Table 5-14
	Applied dose (human equivalent dose)	Mouse, male	Liver		1.0×10^{-2}	Table 5-15
	1995 IRIS assessment	Mouse, male	Liver		7.5×10^{-3}	

[a]GST-T1$^{+/+}$ = homozygous, full enzyme activity; Mixed = genotypes based on a population reflecting the estimated frequency of genotypes in the current U.S. population: 20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).

Bolded value is the basis for the recommended OSF of 1×10^{-3} per mg/kg-day.

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5.4.2. Cancer Inhalation Unit Risk

5.4.2.1. *Choice of Study/Data—with Rationale and Justification*

As discussed in section 4.7, results from several cohort mortality studies of workers repeatedly exposed to dichloromethane and several case-control studies provide some supporting evidence of carcinogenicity in humans, specifically with respect to liver and brain cancer. However, the epidemiologic studies do not provide adequate data to estimate exposure-response relationships for dichloromethane exposure and these cancers.

Results from several bioassays provide sufficient evidence of the carcinogenicity of dichloromethane in mice and rats exposed by inhalation, as well as adequate data to describe dose-response relationships. As discussed in section 4.7.2, repeated inhalation exposure to concentrations of 2,000 or 4,000 ppm dichloromethane produced increased incidences of lung and liver tumors in male and female B6C3F$_1$ mice (Maronpot et al., 1995; Foley et al., 1993; Kari et al., 1993; Mennear et al., 1988; NTP, 1986). A weaker trend was seen with respect to liver tumor incidence (described as neoplastic nodules or hepatic carcinomas) in female rats, but this trend was not seen when limited to hepatic carcinomas (NTP, 1986). A statistically significant increased incidence of brain tumors has not been observed in any of the animal cancer bioassays, but a 2-year study using relatively low exposure levels (0, 50, 200, and 500 ppm) in Sprague-Dawley rats observed a total of six astrocytoma or glioma (mixed glial cell) tumors (combining males and females) in the exposed groups (Nitschke et al., 1988a). These tumors are exceedingly rare in rats, and there are few examples of statistically significant trends in animal bioassays (Sills et al., 1999). Male and female F344 rats exposed by inhalation to 2,000 or 4,000 ppm showed significantly increased incidences of benign mammary tumors (adenomas or fibroadenomas) and the male rats also exhibited a low rate of sarcoma or fibrosarcoma in mammary gland or subcutaneous tissue around the mammary gland (NTP, 1986).

The NTP inhalation study in B6C3F$_1$ mice (NTP, 1986) was used to derive an IUR for dichloromethane because of the completeness of the data, adequate sample size, and clear dose response with respect to liver and lung tumors. The liver tumor incidence in male mice increased from 44% in controls to 66% in the highest dose group; in females the incidence of this tumor rose from 6 to 83%. For lung tumors, the incidence rose from 10 to 80% in males and from 6 to 85% in females. Compelling evidence exists for the role of GST-mediated metabolism of dichloromethane in carcinogenicity in mice (section 4.7.3), and both mice and humans possess this metabolic pathway. Modeling intake, metabolism, and elimination of dichloromethane in mice and humans is feasible. Thus, it is reasonable to apply the best available PBTK models to estimate equivalent internal doses in mice and humans.

The mammary tumor data from the NTP (1986) study was also used to derive a comparative IUR. However, the toxicokinetic or mechanistic events that might lead to mammary gland tumor development in rats are unknown, and so a clear choice of the optimal

internal dose metric could not be made. Thus, this derivation is based on the average daily AUC for dichloromethane in blood. The role of CYP- or GST-mediated metabolism in the mammary gland is uncertain, although both GST-T1 (Lehmann and Wagner, 2008) and CYP2E1 (El-Rayes et al., 2003; Hellmold et al., 1998) expressions have been detected in human mammary tissue. It is also possible that some metabolites enter systemic circulation from the liver and lung, where they are formed.

The female rat liver cancer data from the NTP (1986) inhalation study was not used to derive an IUR because the trend was weaker than that seen in the mouse (incidence increased from 4% in controls to 10% in the highest dose group, trend $p = 0.08$), and because the effect categorization included neoplastic nodule or hepatocellular carcinoma. The brain tumor data seen in the Nitschke et al. (1988a) study in Sprague-Dawley rats were not used to develop an IUR because of the low incidence of this rare tumor (a total of four astrocytoma or glioma tumors in exposed males and two in exposed females). The mechanistic issues with respect to mammary tumors and health effects issues with respect to brain tumors represent data gaps in the understanding of the health effects of dichloromethane and relevance of the rat data to humans.

5.4.2.2. *Derivation of the Cancer Inhalation Unit Risk*

The derivation of the IUR parallels the process described in section 5.4.1.2 for the cancer OSF. Since modeling metabolism and elimination kinetics of dichloromethane in mice and humans is feasible, it is reasonable to apply the best available PBTK models to determine equivalent target organ doses in mice and humans.

5.4.2.3. *Dose-Response Data*

Data for liver and lung tumors in male and female $B6C3F_1$ mice following exposure to airborne dichloromethane were used to develop IURs for dichloromethane (Mennear et al., 1988; NTP, 1986). As discussed in section 5.4.1.8, the liver tumor dose-response data were also the basis of an OSF, derived by route-to-route extrapolation using the PBTK models, to compare with an OSF based on liver tumor data in mice exposed to dichloromethane in drinking water (Serota et al., 1986b). In the NTP (1986) study, significant increases in incidence of liver and lung adenomas and carcinomas were observed in both sexes of $B6C3F_1$ mice exposed 6 hours/day, 5 days/week for 2 years (Table 5-17). Since significant decreases in survival were observed in the treated groups of both sexes, the at-risk study populations (represented by the denominators in the incidence data) were determined by excluding all animals dying prior 52 weeks.

5.4.2.4. *Dose Conversion and Extrapolation Methods: Cancer Inhalation Unit Risk*

Dose conversion

The PBTK model of Marino et al. (2006) for dichloromethane in the mouse was used to simulate inhalation exposures of 6 hours/day, 5 days/week (Mennear et al., 1988; NTP, 1986) and to calculate long-term daily average internal doses. Study-, group-, and sex-specific mean BWs were used. Based on evidence that metabolites of dichloromethane produced via the GST pathway are primarily responsible for dichloromethane carcinogenicity in mouse liver and lung (summarized in section 4.7.3), and the assumption that these metabolites are sufficiently reactive that they do not have substantial distribution outside these tissues, the recommended selected internal dose metric for liver tumors and lung tumors were long-term average daily mass of dichloromethane metabolized via the GST pathway per unit volume of liver and lung, respectively (Table 5-17). Figure 5-16 show the comparison between inhalation external and internal doses in the liver and lung, respectively, using this dose metric for the mouse and for the human. A whole-body metabolism metric was also examined. This metric would be more relevant under a scenario of slowly cleared metabolites that undergo general circulation

Table 5-17. Incidence data for liver and lung tumors and internal doses, based on GST metabolism dose metrics, in male and female B6C3F$_1$ mice exposed to dichloromethane via inhalation for 2 years

Sex, tumor type	BW (g)	External dichloromethane concentration (ppm)	Mouse tumor incidence	Mouse internal tissue dose[a]	Mouse whole body metabolism dose[b]
Male, liver[c]	--	0	22/50 (44%)[e]	0	0
	34.0	2,000	24/47 (51%)	2,363.7	100.2
	32.0	4,000	33/47 (70%)	4,972.2	210.7
Male, lung[d]	--	0	5/50 (10%)[e]	0	0
	34.0	2,000	27/47 (55%)	475.0	100.2
	32.0	4,000	40/47 (85%)	992.2	210.7
Female, liver[c]	--	0	3/47 (6%)[e]	0	0
	30.0	2,000	16/46 (35%)	2,453.2	104.0
	29.0	4,000	40/46 (87%)	5,120.0	217.0

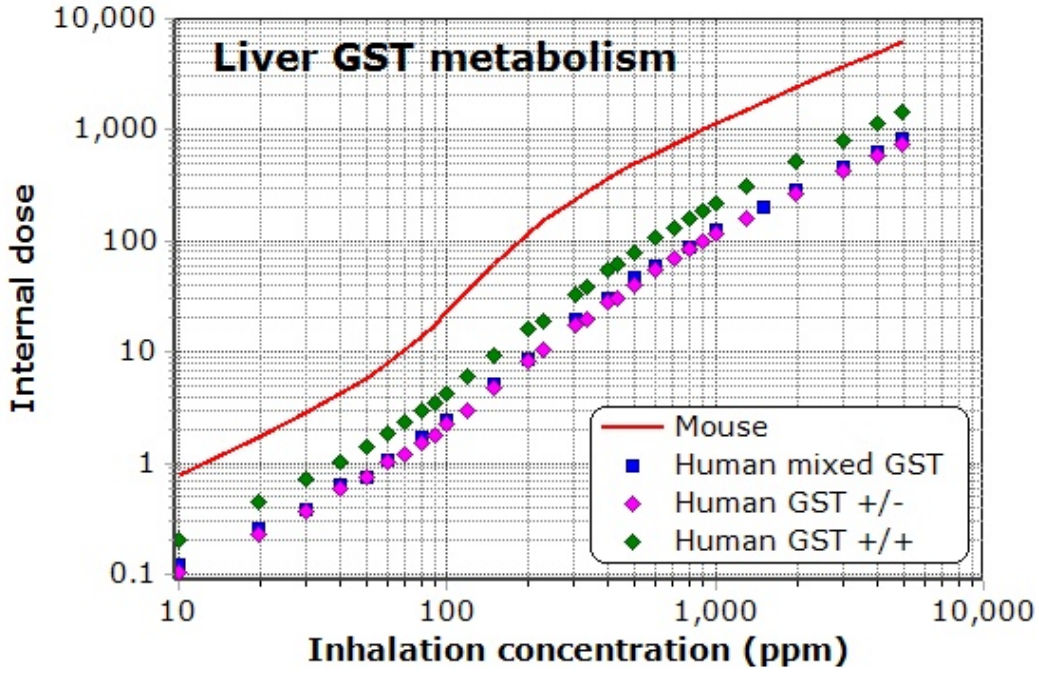
Table 5-17. Incidence data for liver and lung tumors and internal doses, based on GST metabolism dose metrics, in male and female B6C3F$_1$ mice exposed to dichloromethane via inhalation for 2 years

Sex, tumor type	BW (g)	External dichloromethane concentration (ppm)	Mouse tumor incidence	Mouse internal tissue dose[a]	Mouse whole body metabolism dose[b]
Female, lung[d]	--	0	3/45 (6%)[e]	0	0
	30.0	2,000	30/46 (65%)	493.0	104.0
	29.0	4,000	41/46 (89%)	1,021.8	217.0

[a]For liver tumors: mg dichloromethane metabolized via GST pathway/L liver tissue/day from 6 hours per day, 5 days per week exposure; for lung tumors: mg dichloromethane metabolized via GST pathway/L lung tissue/day from 6 hours per day, 5 days per week exposure.
[b]Based on the sum of dichloromethane metabolized via the GST pathway in the lung plus the liver, normalized to total BW (i.e., [lung GST metabolism (mg/day) + liver GST metabolism (mg/day)]/kg BW). Units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day.
[c]Hepatocellular carcinoma or adenoma. Mice dying prior to 52 weeks were excluded from the denominators.
[d]Bronchoalveolar carcinoma or adenoma. Mice dying prior to 52 weeks were excluded from the denominators.
[e]Statistically significant increasing trend (by incidental and life-table tests; $p \leq 0.01$).

Sources: Mennear et al., 1988; NTP, 1986

A.



B.

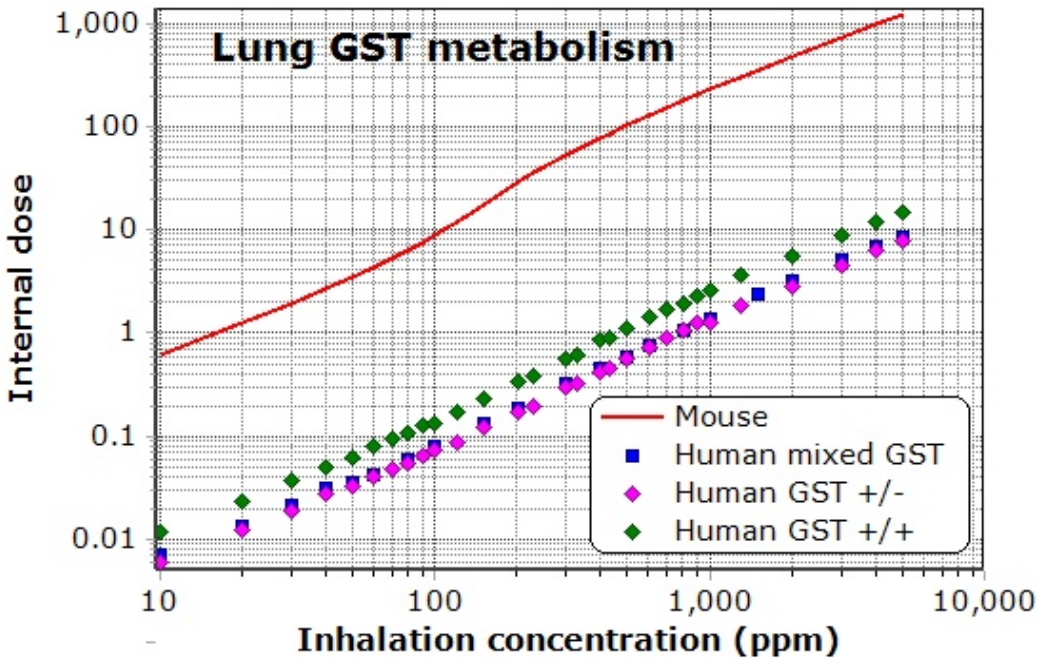


Figure 5-16. PBTK model-derived internal doses (mg dichloromethane metabolized via the GST pathways per liter tissue per day) for liver (A) and lung (B) in mice and humans, and their associated external exposures (ppm), used for the derivation of cancer inhalation unit risks. Average daily doses were calculated from simulated mouse exposures of 6 hours/day, 5 days/week, while simulated human exposures were continuous. The GST metabolism rate in each simulated human population was obtained by generating 1000 random samples from each population (ages 0.5-80 years, males and females) for each exposure level, and calculating the average GST metabolic rate for each sample.

Dose-response modeling and extrapolation

The multistage dose-response model was fit to the mouse tumor incidence and PBTK model-derived internal dose data to derive mouse internal BMD_{10} and $BMDL_{10}$ values associated with 10% extra risk (Table 5-18). Different polynomial models and models dropping dose groups starting with the highest dose group were compared based on adequacy of model fit as assessed by overall χ^2 goodness of fit (p-value > 0.10) and examination of residuals, particularly in the region of the BMR (U.S. EPA, 2000c). Appendix E-2 provides details of the BMD modeling results for the male. The mouse liver and lung tumor risk factors (extra risk per unit internal dose) were calculated by dividing 0.1 by the mouse $BMDL_{10}$ for liver and lung tumors, respectively.

Linear extrapolation from the internal $BMDL_{10}$ ($0.1/BMDL_{10}$) was used to derive inhalation risk factors for lung and liver tumors in male and female mice (Table 5-18). As discussed in section 4.7, the linear low-dose extrapolation approach for agents with a mutagenic mode of action was selected.

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Table 5-18. BMD modeling results and tumor risk factors associated with 10% extra risk for liver and lung tumors in male and female B6C3F$_1$ mice exposed by inhalation to dichloromethane for 2 years, based on liver-specific GST metabolism and whole body GST metabolism dose metrics

Internal dose metric[a]		BMDS model[b]	χ^2 goodness of fit *p*-value	Mouse BMD_{10}[c]	Mouse $BMDL_{10}$[c]	Allometric-scaled human $BMDL_{10}$[d]	Tumor Risk Factor[e]	
							Scaling = 1.0	Allometric-scaled
Liver-specific	Male, liver	MS (0,1)	0.40	913.9	544.4	77.8	1.84×10^{-4}	1.29×10^{-3}
	Male, lung	MS (0,1)	0.64	61.7	48.6	7.0	2.06×10^{-3}	1.44×10^{-2}
	Female, liver	MS (0,2)	0.53	1224.1	659.7	94.2	1.52×10^{-4}	1.06×10^{-3}
	Female, lung	MS (0,1)	0.87	51.2	40.7	5.8	2.46×10^{-3}	1.72×10^{-2}
Whole body	Male, liver	MS (0,1)	0.40	38.7	23.1	3.3	--	3.03×10^{-2}
	Male, lung	MS (0,1)	0.66	13.1	10.3	1.5	--	6.80×10^{-2}
	Female, liver	MS (0,2)	0.53	51.9	28.0	4.0	--	2.50×10^{-2}
	Female, lung	MS (0,1)	0.88	10.8	8.6	1.2	--	8.14×10^{-2}

[a]Liver specific dose units = mg dichloromethane metabolized via GST pathway per liter tissue per day; Whole-body dose units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day)

[b]The multistage (MS) model in EPA BMDS version 2.0 was fit to the mouse dose-response data shown in Table 5-17 using internal dose metrics calculated with the mouse PBTK model. Numbers in parentheses indicate (1) the number of dose groups dropped in order to obtain an adequate fit and (2) the degree polynomial of the model.

[c]BMD_{10} and $BMDL_{10}$ refer to the BMD-model-predicted mouse internal dose and its 95% lower confidence limit, associated with a 10% extra risk for the incidence of tumors.

[d]Mouse $BMDL_{10}$ divided by $(BW_{human}/BW_{mouse})^{0.25} = 7$.

[e]Dichloromethane tumor risk factor (extra risk per unit internal dose) derived by dividing the BMR (0.1) by the mouse $BMDL_{10}$ and by the allometric-scaled human $BMDL_{10}$,, for the scaling = 1.0 and allometric-scaled risk factors, respectively.

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Application of allometric scaling factor

As discussed in section 5.4.1.4., the choice of a scaling factor is based on the question of the role of specific metabolites, and particularly how these metabolites are transformed or removed. If the key metabolite is established and is known to be sufficiently reactive to not spread in systemic circulation, then it can be assumed that 1) the level of reactivity and rate of clearance (i.e., disappearance due to local reactivity) for this metabolite, per volume tissue, is equal in rodents and humans; and (2) risk is proportional to the long-term daily average concentration of the metabolite. Under these assumptions, rodent internal $BMDL_{10}$ values based on tissue-specific dichloromethane metabolism require no allometric scaling to account for toxicodynamic differences and predict the corresponding level of human risk as a function of the metric (i.e., the scaling factor in Figure 5-14 was equal to 1.0). (A single metabolite is referenced, but the same argument holds in general for more than one metabolite). Under this scenario and assumptions, humans and rodents with the same long-term daily average metabolite formation per volume tissue (e.g., equal internal $BMDL_{10}$) should both experience the same long-term average concentration of the metabolite when the metabolite is highly reactive and hence experience the same extra risk. However, some uncertainties remain concerning the hypothesized role of a highly reactive metabolite in the carcinogenic effects seen with dichloromethane. The active metabolite(s) have not been established, and data pertaining to the reactivity or clearance rate of these metabolite(s) are lacking. For example, quantitative measurements of adducts of interest or of the half life of relevant compounds in humans and in mice are not available. To address these uncertainties, use of scaling factor that addresses the possibility that the rate of clearance for the metabolite is limited by processes that are known to scale allometrically, such as blood perfusion or enzyme activity, may be appropriate.. This case would result in use of a mouse:human dose-rate scaling factor of $(BW_{human}/BW_{mouse})^{0.25} = 7$ to adjust the mouse-based $BMDL_{10}$ values downward. Using this internal dose metric (liver-specific metabolism with allometric scaling), equivalent rodent and human internal $BMDL_{10}$ values result in a human liver tumor risk factor ($0.1/BMDL_{10}$) that is assumed equal to that for the mouse, given a 70-year lifetime exposure. Another alternative that can be used is based on an allometrically-scaled whole-body metabolism metric. In this case, less weight is given to the evidence of site-specificity of the effects. As with the OSF derivations, the cancer toxicity values derived using each of these metrics and scaling factors (i.e., liver-specific metabolism with and without allometric-scaling and the whole-body metabolism metric) are presented in the following tables. Considering the lack of data pertaining to clearance rates or the actual AUC of the active carcinogenic metabolite(s) in mice and humans, the IUR recommended by the EPA are based on the allometrically-scaled tissue-specific GST metabolism rate dose metric.

Calculation of IURs

A probabilistic PBTK model for dichloromethane in humans adapted from David et al. (2006) (see Appendix B) was used with Monte Carlo sampling to calculate distributions of internal lung, liver, or blood doses associated with chronic unit inhalation (1 μg/m^3) exposures. The data on which the model is based indicate that relationship between exposure and internal dose is linear at low doses. Parameters in the human PBTK model developed by David et al. (2006) are distributions that incorporate information about dichloromethane toxicokinetic variability and uncertainty among humans. Monte Carlo sampling was performed in which each human model parameter was defined by a value randomly drawn from each respective parameter distribution. The model was then executed by using the external unit exposure as input, and the resulting human equivalent internal dose was recorded. This process was repeated for 10,000 iterations to generate a distribution of human internal doses.

The resulting distribution of IURs shown in Table 5-19 were derived by multiplying the human internal dose tumor risk factor (in units of reciprocal internal dose) by the respective distributions of human average daily internal dose resulting from a chronic unit inhalation exposure of 1 μg/m^3 dichloromethane. Table 5-19 presents the analysis using the male data. Analyses based on the female data produced very similar results, and are summarized in Appendix F. Because adjustments for interindividual variability are not generally used or recommended in cancer risk analysis, the mean slope factor was selected as the recommended value to be used in deterministic risk assessments; other values at the upper end of the distribution are also presented. As with the oral cancer slope factor derivation, the cancer IUR is derived for a population composed entirely of carriers of the GST-T1 homozygous positive genotype (the group that would be expected to be most sensitive to the carcinogenic effects of dichloromethane), and a population reflecting the estimated frequency of GST-T1 genotypes in the current U.S. population (20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$, the "mixed" population). All simulations also included a distribution of CYP activity, based on data from Lipscomb et al. (2003).

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Table 5-19. IURs for dichloromethane based on PBTK model-derived internal liver and lung doses in B6C3F$_1$ male mice exposed via inhalation for 2 years, based on liver-specific GST metabolism and whole body metabolism dose metrics, by population genotype

Internal dose metric and scaling factor[a]	Population genotype[b]	Tumor type	Human tumor risk factor[c]	Distribution of human internal dichloromethane doses from 1 µg/m^3 exposure[d]			Resulting candidate human IUR[e] (µg/m^3)$^{-1}$		
				Mean	95th percentile	99th percentile	Mean	95th percentile	99th percentile
Tissue-specific, allometric-scaled	GST-T1$^{+/+}$	Liver	1.29×10^{-3}	5.64×10^{-6}	1.56×10^{-5}	2.60×10^{-5}	7.3×10^{-9}	2.0×10^{-8}	3.3×10^{-8}
	GST-T1$^{+/+}$	Lung	1.44×10^{-2}	3.31×10^{-7}	8.55×10^{-7}	1.34×10^{-6}	4.8×10^{-9}	1.2×10^{-8}	1.9×10^{-8}
	Mixed	Liver	1.29×10^{-3}	2.62×10^{-6}	8.65×10^{-6}	1.45×10^{-5}	3.4×10^{-9}	1.1×10^{-8}	1.9×10^{-8}
	Mixed	Lung	1.44×10^{-2}	1.81×10^{-7}	5.67×10^{-7}	9.84×10^{-7}	2.6×10^{-9}	8.2×10^{-9}	1.4×10^{-8}
Tissue-specific, scaling = 1.0	GST-T1$^{+/+}$	Liver	1.84×10^{-4}	5.64×10^{-6}	1.56×10^{-5}	2.60×10^{-5}	1.0×10^{-9}	2.9×10^{-9}	4.8×10^{-9}
	GST-T1$^{+/+}$	Lung	2.06×10^{-3}	3.31×10^{-7}	8.55×10^{-7}	1.34×10^{-6}	6.8×10^{-10}	1.8×10^{-9}	2.8×10^{-9}
	Mixed	Liver	1.84×10^{-4}	2.62×10^{-6}	8.65×10^{-6}	1.45×10^{-5}	4.8×10^{-10}	1.6×10^{-9}	2.7×10^{-9}
	Mixed	Lung	2.06×10^{-3}	1.81×10^{-7}	5.67×10^{-7}	9.84×10^{-7}	3.7×10^{-10}	1.2×10^{-9}	2.0×10^{-9}
Whole-body, allometric-scaled	GST-T1$^{+/+}$	Liver	3.03×10^{-2}	1.53×10^{-7}	4.87×10^{-7}	9.20×10^{-7}	4.6×10^{-9}	1.5×10^{-8}	2.8×10^{-8}
	GST-T1$^{+/+}$	Lung	6.80×10^{-2}	1.53×10^{-7}	4.87×10^{-7}	9.20×10^{-7}	1.0×10^{-8}	3.3×10^{-8}	6.3×10^{-8}
	Mixed	Liver	3.03×10^{-2}	8.76×10^{-8}	3.20×10^{-7}	6.76×10^{-7}	2.7×10^{-9}	9.7×10^{-9}	2.1×10^{-8}
	Mixed	Lung	6.80×10^{-2}	8.76×10^{-8}	3.20×10^{-7}	6.76×10^{-7}	6.0×10^{-9}	2.2×10^{-8}	4.6×10^{-8}

[a]Tissue specific dose units = mg dichloromethane metabolized via GST pathway per liter tissue (liver or lung, respectively, for liver and lung tumors) per day; Whole-body dose units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day.

[b]GST-T1$^{+/+}$ = homozygous, full enzyme activity;); mixed = population reflecting estimated frequency of genotypes in current U.S. population: 20% GST-T$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).

[c]Dichloromethane tumor risk factor (extra risk per unit internal dose) derived by dividing the BMR (0.1) by the allometric-scaled human $BMDL_{10}$ or by the mouse $BMDL_{10}$ (from Table 5-18) for the allometric-scaled and scaling = 1.0 risk factors, respectively.

[d]Mean, 95th, and 99th percentile of the human PBTK model-derived probability distribution of daily average internal dichloromethane dose resulting from chronic exposure to 1 µg/m^3 (0.00029 ppm).

[e]Derived by multiplying the dichloromethane tumor risk factor by the PBTK model-derived probabilistic internal doses from daily exposure to 1 µg/m^3

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5.4.2.5. *Cancer Inhalation Unit Risk*

The recommended cancer IURs are $7 \times 10^{-9}(\mu g/m^3)^{-1}$ and $5 \times 10^{-9}(\mu g/m^3)^{-1}$ for the development of liver and lung cancer, respectively, based on the mean for the GST-T1$^{+/+}$ population (the group with the greatest presumed sensitivity). These values are based on male B6C3F$_1$ mice, using a tissue-specific GST metabolism dose metric, with allometric scaling (Table 5-19). Risk estimates were slightly higher for liver tumors and essentially equivalent for lung tumors in males compared to females (Appendix F), so the estimates for males were selected for the candidate values.

Consideration of combined risk (summing risk across tumors)

With two significant tumor sites, focusing on the more sensitive response may underestimate the overall cancer risk associated with exposure to this chemical. Following the recommendations of the National Research Council (NRC, 1994) and the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), an upper bound on total risk was estimated in order to gain some understanding of the total risk from multiple tumor sites in the selected data set. Note that this estimate of overall risk describes the risk of developing either tumor type, not just the risk of developing both simultaneously.

NRC (1994) stated that an approach based on counts of animals with one or more tumors (or tumor-bearing animals) would tend to underestimate overall risk when tumor types occur independently and that an approach based on combining the risk estimates from each separate tumor type should be used. For dichloromethane, there is no reason to expect that the occurrence of one tumor type depends on the incidence of the other, given the association of different dose metrics with each tumor response. Therefore it appears reasonable to assume that the two tumor types occur independently. However, simply summing upper limit risks may result in an overestimation of overall of combined risk because of the statistical issues with respect to summing variances of distributions. An additional challenge results from the use of different internal dose metrics for different tumors, as is the case with the dose metrics based on tissue-specific metabolism. Statistical methods based on a common metric can not be used with the tissue-specific metabolism metric used in these derivations.

An alternative approach is to derive an upper bound on the combined risk estimates by summing central tendency risks and calculating a pooled SD by using BMD_{10} and $BMDL_{10}$ values for liver and lung tumors. The SD associated with the IUR for each tumor site is calculated as the difference between 95th percentiles of the distribution for upper bound and maximum likelihood estimate IURs (based on either female or male mouse tumor risk factors), divided by 1.645 (the relevant *t* statistic, assuming normal distributions of summed quantities). Variances for each tumor site are the squares of the SDs. Pooled variance and SD are defined as the sum of variances for lung and liver tumors and the square root of that sum, respectively. Finally, the upper bound on the combined lung and liver cancer risk is determined by multiplying

the cumulative SD by 1.645 and adding it to the summed central tendency IURs. The calculations of these upper bound estimates for combined liver and lung tumor risks are shown in Table 5-20.

Using this approach and the male mouse-derived risk factors, the combined human equivalent IUR values for both tumor types is 1×10^{-8} $(\mu g/m^3)^{-1}$ (rounded from 1.1×10^{-8}) in the most sensitive (GST-$T1^{+/+}$) population. This is the recommended inhalation cancer unit risk value to be used in deterministic risk assessments for chronic exposure to dichloromethane. The corresponding value for a population with the frequency distribution of GST-T1 genotypes currently found in the U.S. population is 6×10^{-9} $(\mu g/m^3)^{-1}$.

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Table 5-20. Upper bound estimates of combined human IURs for liver and lung tumors resulting from lifetime exposure to 1 µg/m^3 dichloromethane, based on liver-specific GST metabolism and whole body metabolism dose metrics, by population genotype

Internal dose metric and scaling factor[a]	Population genotype[b]	Tumor site	Upper bound IUR[c]	Central tendency IUR[d]	Variance of tissue-specific tumor risk[e]	Combined tumor risk SD[f]	Upper bound on combined tumor risk[g] (µg/m^3)$^{-1}$
Tissue-specific, allometric-scaled	GST-T1$^{+/+}$	Liver	7.3×10^{-9}	4.3×10^{-9}	3.17×10^{-18}		
		Lung	4.8×10^{-9}	3.8×10^{-9}	3.75×10^{-19}		
		Liver or lung		8.1×10^{-9}		1.9×10^{-9}	1.1×10^{-8}
	Mixed	Liver	3.4×10^{-9}	2.0×10^{-9}	6.84×10^{-19}		
		Lung	2.6×10^{-9}	2.1×10^{-9}	1.12×10^{-19}		
		Liver or lung		4.1×10^{-9}		8.9×10^{-10}	5.5×10^{-9}
Tissue-specific, scaling = 1.0	GST-T1$^{+/+}$	Liver	1.0×10^{-9}	6.2×10^{-10}	6.48×10^{-20}		
		Lung	6.8×10^{-10}	5.4×10^{-10}	7.72×10^{-21}		
		Liver or lung		1.2×10^{-9}		2.7×10^{-10}	1.6×10^{-9}
	Mixed	Liver	4.8×10^{-10}	2.9×10^{-10}	1.40×10^{-20}		
		Lung	3.7×10^{-10}	2.9×10^{-10}	2.31×10^{-21}		
		Liver or lung		5.8×10^{-10}		1.3×10^{-10}	7.9×10^{-10}
Whole-body, allometric-scaled	GST-T1$^{+/+}$	Liver	4.6×10^{-9}	2.8×10^{-9}	1.29×10^{-18}		
		Lung	1.0×10^{-8}	8.2×10^{-9}	1.84×10^{-18}		
		Liver or lung		1.1×10^{-8}		1.8×10^{-9}	1.4×10^{-8}
	Mixed	Liver	2.7×10^{-9}	1.6×10^{-9}	4.23×10^{-19}		
		Lung	6.0×10^{-9}	4.7×10^{-9}	6.04×10^{-19}		
		Liver or lung		6.3×10^{-9}		1.0×10^{-9}	7.9×10^{-9}

[a]Tissue specific dose units = mg dichloromethane metabolized via GST pathway per liter tissue (liver or lung, respectively, for liver and lung tumors) per day; Whole-body dose units = mg dichloromethane metabolized via GST pathway in lung and liver/kg-day.
[b]GST-T1$^{+/+}$ = homozygous, full enzyme activity;); mixed = population reflecting estimated frequency of genotypes in current U.S. population: 20% GST-T$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).
[c]Estimated at the human equivalent $BMDL_{10}$ ($0.1/BMDL_{10}$) (see Table 5-18).
[d]Estimated at the human equivalent BMD_{10} (0.1/BMD) (see Table 5-18).
[e]Calculated as the square of the difference of the upper bound and central tendency IURs divided by the *t* statistic, 1.645.
[r]Calculated as the square root of the sum of the variances for liver and lung tumors.
[g]Calculated as the product of the cumulative tumor risk SD and the *t* statistic, 1.645, added to the sum of central tendency IURs.

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5.4.2.6. Comparative Derivation Based on Rat Mammary Tumor Data

Mammary gland tumor data from male and female F344 rats following an inhalation exposure to dichloromethane were considered in development of a comparative IUR for dichloromethane (Mennear et al., 1988; NTP, 1986). In both the male and female rats, there were significant increases in the incidence of adenomas, fibroadenomas, or fibromas in or near the mammary gland. These were characterized as benign tumors in the NTP report (NTP, 1986). Increased numbers of benign mammary tumors per animal in exposed groups were also seen in two studies of Sprague-Dawley rats (Nitschke et al., 1988a; Burek et al., 1984). An oral (gavage) study in Sprague-Dawley rats reported an increased incidence of malignant mammary tumors, mainly adenocarcinomas (8, 6, and 18% in the control, 100, and 500 mg/kg dose groups, respectively), but the increase was not statistically significant. Data were not provided to allow an analysis that accounts for differing mortality rates (Maltoni et al., 1988). There are considerably more uncertainties regarding the interpretation of these data with respect to carcinogenic risk compared with the data pertaining to liver and lung tumors. The trends were driven in large part by benign tumors; adenocarcinomas and carcinomas were seen only in the females, with incidences of 1, 2, 2, and 0 in the 0, 1000, 2000 and 4000 ppm exposure groups, respectively. There are little data to guide the choice of relevant dose metric, and the genotoxicity and mechanistic studies have not included mammary tissue. For these reasons, the analysis and the calculation of the comparative IUR based on rat mammary tumor data are presented in Appendix G. The IUR based on the female rat data was 1×10^{-7} $(\mu g/m^3)^{-1}$.

5.4.2.7. *Alternative Based on Administered Concentration*

Another comparison that can be made is with an alternative IUR based on liver and lung tumors in mice, using the external concentrations of dichloromethane in the mouse studies as converted to HECs, and then applying this using BMD modeling to obtain the $BMDL_{10}$ and resulting IUR. Mouse bioassay exposures were adjusted to HECs as follows:

- Adjusting to continuous exposure: External $concentration_{ADJ}$ = External concentration × (6 hours/24 hours) × (5 days/7 days)
- Concentrations in mg/m^3 = concentrations in ppm × 84.93/24.45.
- $[H_{b/g}]_A/[H_{b/g}]_H$ = the ratio of blood:gas (air) partition coefficients in animals and humans. Because the partition coefficient for mice (23.0) is higher than for humans (9.7), a value of 1.0 was used, as per U.S. EPA (1994b) guidance.

Thus,

HECs = External $concentration_{ADJ}$ × $[H_{b/g}]_A/[H_{b/g}]_H$ = External $concentration_{ADJ}$ × 1

The HECs (mg/m^3) for the 0, 2,000, and 4,000 ppm exposure groups were 0, 1,241, and 2,481 mg/m^3, respectively. The BMD modeling and IURs derived from these values, in conjunction with the liver and lung tumor data from Table 5-17 (NTP, 1986), are shown in Table 5-21. The resulting IURs, based on the liver or lung tumors in the mouse, are approximately one order of magnitude higher than the currently recommended value obtained by using the mouse and human PBTK models.

Table 5-21. Inhalation units risks based on human $BMDL_{10}$ values using administered concentration for liver and lung tumors in $B6C3F_1$ mice exposed by inhalation to dichloromethane for 2 years

Sex, tumor type	BMDS model[a]	χ^2 goodness of fit *p*-value	BMD_{10}[b]	$BMDL_{10}$[b]	Inhalation unit risk[c] $(\mu g/m^3)^{-1}$
Male, liver	MS (0,1)	0.44	456.17	272.01	3.7×10^{-7}
Male, lung	MS (0,1)	0.21	137.51	108.27	9.2×10^{-7}
Female, liver	MS (0,2)	0.37	602.67	344.26	2.9×10^{-7}
Female, lung	MS (0,1)	0.76	126.68	100.78	9.9×10^{-7}

[a]The multistage (MS) model in EPA BMDS version 2.0 was fit to each of the four sets of mouse dose-response data shown in Table 5-17. The HEC used in these models for the 0, 2,000,and 4,000 ppm exposure groups were 0, 1,241, and 2,481 mg/m^3, respectively. Numbers in parentheses indicate (1) the number of dose groups dropped in order to obtain an adequate fit, and (2) the lowest degree polynomial of the model showing an adequate fit.
[c]BMD_{10} and $BMDL_{10}$ refer to the BMD-model-predicted HECs (mg dichloromethane per cubic meter), and its 95% lower confidence limit, associated with a 10% extra risk for the incidence of tumors.
[d]IUR, (risk/mg-m^3) = 0.1/human $BMDL_{10}$.

Sources: Mennear et al. (1988); NTP (1986).

The difference between the administered concentration methodology and PBTK-based approaches depends on two key differences: the use of a dichloromethane-metabolite dose-metric, rather than dichloromethane AUC, and the fact that the rate of dichloromethane conversion to that metabolite is estimated in humans by using human data rather than default allometric scaling ($BW^{0.75}$). In addition, the administered concentration methodology does not account in any way for the GST polymorphism and so might be considered as representing the general/mixed-GST-genotype population rather than the +/+ subpopulation.

5.4.2.8. *Previous IRIS Assessment: Cancer Inhalation Unit Risk*

The IUR in the previous IRIS assessment was determined from the combined incidence of liver and lung adenomas and carcinomas in $B6C3F_1$ mice exposed to dichloromethane for 2 years by NTP (1986). A value of 4.7×10^{-7} $(\mu g/m^3)^{-1}$ was derived by the application of a modified version of the PBTK model of Andersen et al. (1987), which incorporated the

pharmacokinetics and metabolism of dichloromethane. Internal dose estimates, based on dichloromethane metabolism via the GST pathway, were used and corrected for differences in interspecies sensitivity by applying an interspecies scaling factor of 12.7, which was based on dose equivalence adjusted to BW to the 2/3 power, to the human risks (Rhomberg, 1995; U.S. EPA, 1987a).

5.4.2.8. *Comparison of Cancer Inhalation Unit Risk Using Different Methodologies*

In this assessment, cancer IURs derived by using different dose metrics and assumptions were examined, as summarized in Table 5-22. The recommended IUR value of 1×10^{-8} $(\mu g/m^3)^{-1}$ is based on a tissue-specific GST-internal dose metric, with allometric scaling, because of the evidence for the involvement of highly reactive metabolites formed via the GST pathway. The value derived specifically for the GST-$T1^{+/+}$ population is recommended to provide protection for the population that is hypothesized to be most sensitive to the carcinogenic effect. The values based on the GST-$T1^{+/+}$ group are approximately two to fivefold higher than those for the full population, for all dose metrics used in this assessment. Within a genotype population, the values of the IUR among the various dose metrics vary by about one to two orders of magnitude.

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Table 5-22. Comparison of IURs derived by using various assumptions and metrics

Population[a]	Dose metric	Species, sex,	Tumor type	Scaling factor	IUR[b] $(\mu g/m^3)^{-1}$	Source (Table)
GST-T1$^{+/+}$	**Tissue-specific GST-metabolism rate[c]**	**Mouse, male**	**Liver and lung**	**7.0**	**1.1×10^{-8}**	**Table 5-20**
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver	7.0	7.3×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Lung	7.0	4.8×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	1.0	1.6×10^{-9}	Table 5-20
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	1.0×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Lung	1.0	6.8×10^{-10}	Table 5-19
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Liver and lung	7.0	1.4×10^{-8}	Table 5-20
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	4.6×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Lung	7.0	1.0×10^{-8}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	7.0	5.5×10^{-9}	Table 5-20
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver	7.0	3.4×10^{-9}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Lung	7.0	2.6×10^{-9}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	1.0	7.9×10^{-10}	Table 5-20
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	4.8×10^{-10}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Lung	1.0	3.7×10^{-10}	Table 5-19
Mixed	Whole-body GST metabolism rate	Mouse, male	Liver and lung	7.0	7.9×10^{-9}	Table 5-20
Mixed	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	2.7×10^{-9}	Table 5-19
Mixed	Whole-body GST metabolism rate	Mouse, male	Lung	7.0	6.0×10^{-9}	Table 5-19
	Administered concentration (HEC)	Mouse, male	Liver		3.7×10^{-7}	Table 5-21
	Administered concentration (HEC)	Mouse, male	Lung		9.2×10^{-7}	Table 5-21
	1995 IRIS assessment[c]	Mouse, male	Liver and lung	12.7	4.7×10^{-7}	

[a]GST-T1$^{+/+}$ = homozygous, full enzyme activity; Mixed = genotypes based on a population reflecting the estimated frequency of genotypes in the current U.S. population: 20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).
[b]Based on mean value of the derived distributions
Bolded value is the basis for the recommended IUR of 1×10^{-8} µg/m^3 per mg/kg-day.

5.4.3. Differences Between Current Assessment and Previous IRIS PBTK-based Assessment

To better understand the changes in assessment risk predictions between previous EPA evaluations and the current assessment, the differences in PBTK model parameters for the $B6C3F_1$ mouse were evaluated. Values that differed significantly between the model version used previously and that of Marino et al. (2006), along with derived group parameters that lend further insight, are shown in Table 5-23.

The tissue:air partition coefficients in Table 5-23 show that, while several of the blood:air partition coefficients appear to differ significantly between the two models, the corresponding blood:air partition coefficient does not, and it is the latter that can be more indicative of long-term equilibration between the tissue (tissue group) and air. Thus, the partition coefficients, the ones that most significantly differ are the blood:air and liver:air partition coefficients that are, respectively, 2.8- and 2.6-fold higher in the current version. The increased blood:air partition coefficient results in a tendency for simulated blood concentrations to rise more quickly and reach higher values, other parameters being equal. The significantly increased QCC and VPR contribute even more to this difference, resulting in an even faster rise to steady state during inhalation exposure simulations, though also more rapid delivery to the liver (decreasing blood-flow limitation of hepatic metabolism) and more rapid exhalation. The increased liver:air partition coefficient leads to higher predicted liver concentrations (again, other parameters being equal) and hence higher rates of metabolism.

For metabolism, a much reduced oxidative metabolism is seen, which at low doses depends on $V_{max}c/K_m$. The revised hepatic metabolism is over 40% lower and the total of lung + liver metabolism is 50% lower than previously used. This lower rate of metabolism means that far less of parent dichloromethane will be removed through metabolism, and hence predicted blood concentrations will be higher still, relative to the impact of changes in partition coefficient, QCC, and VPR, as noted above.

The result of having higher predicted blood and liver dichloromethane concentrations is that, while the GSH-pathway metabolic constant, k_fC , is virtually the same for the mouse liver in both cases, the much higher concentration of dichloromethane available will lead to a much higher predicted rate of metabolism via this pathway. For the lung, since the lung:liver ratio (A2) is 43% higher in the model of Marino et al. (2006), the relative increase will be even greater.

Because the revised rate of GST metabolism in mice was estimated by using data with CYP2E1 inhibited by a suicide inhibitor, there is considerable confidence in the relative rate of metabolism by these two pathways, and the GST pathway in particular. The partition coefficients used by Marino et al. (2006) are as measured by Clewell et al. (1993) and expected to be at least as reliable as those used in the 1995 assessment. Considering that the revised PBTK model does an excellent job of reproducing closed-chamber gas uptake data that were not

available for calibration of the 1987 model, as well blood concentrations after intravenous injection, we can have fairly high confidence in its predictions.

The net result of these model changes is that, under mouse bioassay conditions, the predicted dose metrics for liver and lung cancer, i.e., GST-mediated metabolism, are higher than those obtained with the previous model, resulting in a lower risk estimated per unit of dose.

Table 5-23. Comparison of key B6C3F$_1$ mouse parameters differing between prior and current PBTK model application

Parameter[a]	Marino et al. (2006); mean values as applied (posterior)	U.S. EPA (1988b, 1987a, b)
Partition coefficients		
PB blood/air	23	8.29
PF fat/blood	5.1	14.5
PF·PB (fat/blood)·(blood/air) = fat/air	117.3	120.2
PL liver/blood	1.6	1.71
PL·PB (liver/blood)·(blood/air) = liver/air	36.8	14.2
PLu lung (tissue)/blood	0.46	1.71
PLu·PB (lung/blood)·(blood/air) = lung/air	10.6	14.2
PR rapidly perfused/blood	0.52	1.71
PR·PB rapidly perfused/air	12.0	14.2
PS slowly perfused/blood	0.44	0.96
PS·PB slowly perfused/air	10.1	7.96
Flow rates		
QCC cardiac output (L/hour/kg$^{0.74}$)	24.2	14.3
VPR ventilation:perfusion ratio	1.45	1.0
Metabolism parameters		
$V_{max}c$ maximum CYP metabolic rate (mg/hour/kg$^{0.7}$)	9.27	11.1
K_m CYP affinity (mg/L)	0.574	0.396
V_{maxC}/K_m (L/hour/kg$^{0.7}$)	16.1	28
A1 ratio of lung $V_{max}c$ to liver $V_{max}c$	0.207	0.416
Total lung + liver $V_{max}c/K_m$	19.5	39.7
k_fC first-order GST metabolic rate constant (kg$^{0.3}$/hr)	1.41	1.46
	0.196	0.137
A2 ratio of lung k_fC to liver k_fC	1.69	1.66
Total lung + liver k_fC		

[a]Parameters not listed differed by less than 10% between versions. See Table 3-5 and associated text for details.

The other piece of the PBTK-based risk estimation is the human model. In updating the parameter estimates for the human model (see Appendix B for details), the oxidative metabolism $V_{max}c/K_m$ approximately doubled, which leads to lower predicted blood concentrations of dichloromethane available for metabolism by GST. In addition, the liver GST was reduced by almost 60%, and the lung:liver GST ratio decreased by almost fivefold, for a net change in lung

GST of over 90%. Given the larger human data set available to David et al. (2006) and the sophisticated Bayesian analysis used to recalibrate the model, the expectation is that these values are more reliable than the values used in the 1995 IRIS assessment.

Since actual rates of metabolism at a given exposure level also depend on respiration rate and blood flows, these changes in metabolic parameters do not completely determine the relative (predicted) dosimetry. But the difference in cancer risk predictions between the current and previous assessments is primarily explained by the overall prediction of higher GST-mediated dosimetry in the mouse (at bioassay conditions) and lower human GST metabolism (due in part to greater predicted clearance of dichloromethane by oxidative metabolism). In addition to these changes in PBTK parameters, the reduction of scaling factor from 12.7 to 7 is a significant factor in the overall change from the previous assessments.

5.4.4. Application of Age-Dependent Adjustment Factors (ADAFs)

The available dichloromethane studies do not include an evaluation of early-life susceptibility to dichloromethane cancer risk. In the absence of this type of data and if a chemical follows a mutagenic mode of action for carcinogenicity, like dichloromethane, the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA, 2005b) recommends that ADAFs be applied to the cancer values. Since the OSF of 1×10^{-3} $(mg/kg\text{-}day)^{-1}$ and the IUR of 1×10^{-8} $(\mu g/m^3)^{-1}$ were calculated from adult dichloromethane exposures, early-life cancer susceptibility has not been accounted for in these values and ADAFs need to be applied in combination with exposure informations when estimating cancer risks that include early-life exposures. Sample calculations that incorporate ADAFs into the cancer risks are presented in subsequent sections. Additional examples of evaluations of cancer risks incorporating early-life exposure are provided in section 6 of the *Supplemental Guidance* (U.S. EPA, 2005b).

In the *Supplemental Guidance* (U.S. EPA, 2005b), ADAFs are established for three age groups. An ADAF of 10 is applied for age groups less than 2 years, 3 is applied for ages 2 to <16 years, and 1 is applied for 16 years and above (U.S. EPA, 2005b). The 10- and 3-fold adjustments in cancer values are combined with age-specific exposure estimates, when early-life exposure considerations need to be included in cancer risk estimates. The most current information on usage of ADAFs can be found at www.epa.gov/cancerguidelines. For estimation of risk, the *Supplemental Guidance* (U.S. EPA, 2005b) recommends obtaining and using age-specific values for exposure and cancer potency. In the absence of age-specific values cancer potency values, as is true for dichloromethane, age-specific cancer values are estimated by using the appropriate ADAFs. Using this process, a cancer risk is derived for each age group. The risks are summed across the age groups to get the total cancer risk for the age-exposure period of interest.

8329
8330 **5.4.4.1.** ***Application of ADAFs in Oral Exposure Scenarios***
8331 Sample calculations incorporating the use of ADAFs are presented for three exposure
8332 duration scenarios. These scenarios include full lifetime exposure (assuming a 70-year lifespan),
8333 and two 30-year exposures from ages 0–30 and ages 20–50. A constant dichloromethane
8334 exposure of 1 mg/kg-day was assumed for each scenario.
8335 Table 5-24 lists the four factors (ADAFs, OSF, assumed exposure, and duration
8336 adjustment) that are needed to calculate the partial cancer risk based on the early age-specific
8337 group. The partial cancer risk for each age group is the product of the four factors in columns 2–
8338 5. Therefore, the partial cancer risk following daily dichloromethane oral exposure in the age
8339 group 0 to <2 years is the product of the values in columns 2–5 or $10 \times (1 \times 10^{-3}) \times 1 \times 2/70 = 2.9 \times 10^{-4}$. The partial risks that are listed in the last column of Table 5-24 are added together to
8341 get the total risk. Thus, a 70-year (lifetime) risk estimate for continuous exposure to 1 mg/kg-
8342 day dichloromethane is 1.7×10^{-3} per mg/kg-day, which is adjusted for early-life susceptibility
8343 and assumes a 70-year lifetime and constant exposure across age groups.
8344

Table 5-24. Application of ADAFs to dichloromethane cancer risk following a lifetime (70-year) oral exposure

Age group (years)	ADAF	Unit risk (per mg/kg-day)	Exposure concentration (mg/kg-day)	Duration adjustment	Partial risk
0 – <2	10	1×10^{-3}	1	2 years/ 70 years	2.9×10^{-4}
2 – <16	3	1×10^{-3}	1	14 years/ 70 years	6.0×10^{-4}
≥16	1	1×10^{-3}	1	54 years/ 70 years	7.7×10^{-4}
				Total risk	**1.7×10^{-3}**

8345
8346
8347 In calculating the cancer risk for a 30-year constant exposure to dichloromethane at an
8348 exposure level of 1 mg/kg-day from ages 0–30, the duration adjustments would be 2/70, 14/70,
8349 and 14/70 and the partial risks for the three age groups would be 2.9×10^{-4}, 6.0×10^{-4}, and 2.0×10^{-4}, which would result in a total risk estimate of 1.1×10^{-3}.
8351 In calculating the cancer risk for a 30-year constant exposure to dichloromethane at an
8352 exposure level of 1 mg/kg-day from ages 20–50, the duration adjustments would be 0/70, 0/70,
8353 and 30/70. The partial risks for the three groups are 0, 0, and 4.3×10^{-4}, which would result in a
8354 total risk estimate of 4.3×10^{-4}.
8355
8356 **5.4.4.2.** ***Application of ADAFs in Inhalation Exposure Scenarios***

Sample calculations incorporating the use of ADAFs are presented for three exposure duration scenarios involving inhalation exposure. These scenarios include full lifetime exposure (assuming a 70-year lifespan) and two 30-year exposures from ages 0–30 and ages 20–50. A constant dichloromethane inhalation exposure of 1 $\mu g/m^3$ was assumed for each scenario.

Similar to the oral exposure scenarios presented in section 5.4.4.1, Table 5-25 lists the four factors (ADAFs, unit risk, assumed exposure, and duration adjustment) that are needed to calculate the partial cancer risk based on the early age-specific group. The partial cancer risk for each age group is the product of the four factors in columns 2–5. Therefore, the partial cancer risk following daily dichloromethane inhalation exposure in the age group 0 to <2 years is the product of the values in columns 2–5 or $10 \times (1 \times 10^{-8}) \times 1 \times 2/70 = 2.9 \times 10^{-9}$. The partial risks that are listed in the last column of Table 5-25 are added together to get the total risk. Thus, a 70-year (lifetime) risk estimate for continuous exposure to 1 $\mu g/m^3$ dichloromethane is 1.8×10^{-8} per $\mu g/m^3$, which is adjusted for early-life susceptibility and assumes a 70-year lifetime and constant exposure across age groups.

Table 5-25. Application of ADAFs to dichloromethane cancer risk following a lifetime (70-year) inhalation exposure

Age group (years)	ADAF	Unit risk (per mg/kg-day)	Exposure concentration (mg/kg-day)	Duration adjustment	Partial risk
0 – <2	10	1×10^{-8}	1	2 years/ 70 years	2.9×10^{-9}
2 – <16	3	1×10^{-8}	1	14 years/ 70 years	6.0×10^{-9}
≥16	1	1×10^{-8}	1	54 years/ 70 years	7.7×10^{-9}
				Total risk	**1.7×10^{-8}**

In calculating the cancer risk for a 30-year constant exposure to dichloromethane at a level of 1 $\mu g/m^3$ from ages 0–30, the duration adjustments would be 2/70, 14/70, and 14/70, and the partial risks for the three age groups are 2.9×10^{-9}, 6.0×10^{-9}, and 2.0×10^{-9}. These partial risks result in a total risk estimate of 1.1×10^{-8}.

In calculating the cancer risk for a 30-year constant exposure to dichloromethane at a level of 1 $\mu g/m^3$ from ages 20–50, the duration adjustments would be 0/70, 0/70, and 30/70, and the partial risks for the three age groups are 0, 0, and 4.3×10^{-9}, resulting in a total risk estimate of 4.3×10^{-9}.

5.4.5. Uncertainties in Cancer Risk Values

The derivation of cancer risk values often involves a number of uncertainties in the extrapolation of dose-response data in animals to cancer risks in human populations. Several types of uncertainty have been quantitatively integrated into the derivation of the recommended

OSFs and IURs for dichloromethane, while others are qualitatively considered. Table 5-26 and the ensuing discussion summarize the principal uncertainties identified, their possible effects on the cancer risk values, decisions made in the derivations, and justifications for the decisions.

Table 5-26. Summary of uncertainty in the derivation of cancer risk values for dichloromethane

Consideration/ approach	Impact on cancer risk value	Decision	Justification
Selection of data set	Selection of an alternative data set or target organ could change the recommended cancer risk values.	Select Serota et al. (1986b) and NTP (1986) as principal studies to derive recommended liver and lung cancer risks for humans from responses in mice.	The NTP (1986) inhalation bioassay with mice provides the strongest cancer responses (liver and lung tumors) and the best dose-response data in the animal database. The Serota et al. (1986b) mouse drinking water study provides the best oral dose-response data for liver tumors. Dichloromethane carcinogenicity appears to be mediated by a metabolic pathway that is also present in humans (i.e., the GST pathway). In combination with the animal results, epidemiological studies provide evidence of increased risks for liver and biliary duct tract cancer but are limited by a number of factors discussed in sections 4.1.3.6 and 4.1.3.7.
Selection of target organ	Selection of a target organ could change the recommended cancer risk values.	Examine cancer risk values based on alternative tumor responses (mammary gland tumors in rats); identification of potential brain cancer risk as a data gap.	Inhalation cancer risk values based on mammary tumors in rats are about one order of magnitude higher than risk values based on liver or lung tumors in mice, but the evidence for mammary gland tumors from dichloromethane exposure is less consistent than evidence for liver and lung tumors.
Selection of extrapolation approach	Selection of extrapolation approach could change the recommended cancer risk values.	Examine cancer risk values based on alternative approaches.	Oral cancer risk values based on route-to-route extrapolation from the NTP (1986) inhalation mouse bioassay were about one order of magnitude lower than values based on liver tumors in orally exposed mice (Serota et al., 1986b) (see Table 5-16) but are inherently less certain than the values based on oral exposure due to the influence of route of exposure on toxicokinetics.
Selection of dose metric	Selection of dose metric could change the recommended cancer risk values.	Evidence of GST involvement supports focus on this pathway. Cancer risk estimates based on alternative (tissue-specific versus whole-body) metrics examined.	Inhalation and oral liver cancer risk values derived using a tissue-specific GST metabolism dose metric were slightly higher than values derived using a whole-body GST metabolism dose metric; for lung tumors, the reverse pattern is seen. The values based on liver or lung tumors using the tissue-specific GST metabolism are recommended based on the evidence of site locality of effects

Table 5-26. Summary of uncertainty in the derivation of cancer risk values for dichloromethane

Consideration/ approach	Impact on cancer risk value	Decision	Justification
Dose-response modeling	Human risk values could increase or decrease, depending on fits of alternative models	Use multistage dose-response model to derive a BMD.	The multistage model has biological support and is the model most consistently used in EPA cancer assessments.
Low-dose extrapolation	Human risk values would be expected to decrease with the application of nonlinear tumor responses in low-dose regions of dose-response curves.	Use linear extrapolation of risk in low-dose region.	Linear extrapolation from the human tumor risk factors was used to derive cancer risk values for oral and inhalation exposures. The linear low-dose extrapolation approach for agents with a mutagenic mode of action was selected.
Interspecies extrapolation of dosimetry and risk	Alternative values for PBTK model parameters and cross-species scaling factor could increase or decrease human cancer risk values.	Use PBTK model and allometric scaling for the primary dose metric.	Application of rodent and human PBTK models reduced uncertainty on cancer risk values due to interspecies differences in toxicokinetics. Examination of impact of different values for key parameters in human model, and sensitivity analysis of rodent PBTK model parameters identified influential metabolic parameters for which little or no experimental data exist (see *Interspecies Extrapolation of Dosimetry of Risk* section, below).
Sensitive subpopulations	Differences in CYP and GST metabolic rates could change cancer risk values.	CYP variability incorporated in the PBTK model; separate risk estimates generated for the presumed most sensitive (GST-T1$^{+/+}$) genotype	No data are available to determine the range of human toxicodynamic variability/sensitivity, including whether children are more sensitive than adults. The toxicokinetic effect of the GST-T1 polymorphism is included in the human PBTK model, as are other sources of variability in GST and CYP metabolic parameters.

Data selections for derivation of IUR and OSF

The database of animal bioassays identifies the liver and lung as the most sensitive target organs for dichloromethane-induced tumor development. These effects demonstrate a dose-response relationship in mice exposed orally (liver only) or by inhalation (liver and lung). Statistically significant increases in benign mammary gland tumors were observed in one study of F344 rats exposed by inhalation to 2,000 or 4,000 ppm (Mennear et al., 1988; NTP, 1986), and evidence for a tumorigenic mammary gland response in Sprague-Dawley rats was limited to increased numbers of benign mammary tumors per animal at levels of 50–500 ppm (Nitschke et al., 1988a) or 500–3,500 ppm (Burek et al., 1984). An oral (gavage) study in female Sprague-Dawley rats reported an increased incidence of malignant mammary tumors, mainly adenocarcinomas (8, 6, and 18% in the control, 100, and 500 mg/kg dose groups, respectively),

but the increase was not statistically significant. Data were not provided to allow an analysis that accounts for differing mortality rates (Maltoni et al., 1988). The toxicokinetic or mechanistic events that might lead to mammary gland tumor development in rats are unknown, although CYP2E1 (El-Rayes et al., 2003; Hellmold et al., 1998) and GST-T1 expression has been detected in human mammary tissue (Lehmann and Wagner, 2008). Rare CNS tumors were observed in one study in rats, a study spanning a relatively low range of exposures (0–500 ppm). These cancers were not seen in two other studies (NTP, 1986; Burek et al., 1984) in rats, both involving higher doses (1,000–4,000 ppm), or in a similar high-dose study (NTP, 1986) in mice. The relative rarity of the tumors precludes the use of the low-dose exposure study in a quantitative dose-response assessment. The in vivo genotoxicity and mechanistic data in rodents provide a detailed sequence of steps from generation of reactive metabolites to mutagenic effects, such as DNA-protein cross-links and DNA strand breaks. Further, the toxicokinetic pathways implicated in production of the putative carcinogenic metabolites in animals also exist in humans. Thus, there is high confidence that the dose-response data for liver and lung cancer in mice represents the best data currently available for derivation of human cancer risks. A more complete understanding of the carcinogenic potential of dichloromethane would be achieved by addressing data gaps identified with respect to issues regarding potential risk and mechanisms relating to brain cancer and mammary tumors. The available epidemiologic studies provide some evidence of an association between dichloromethane and brain cancer (see Section 4.1.3.7.1). The available epidemiologic studies do not provide an adequate basis for the evaluation of the role of dichloromethane in breast cancer because there are currently no cohort studies with adequate statistical power and no case-control studies with adequate exposure methodology to examine this relationship (see section 4.1.3.7.6)

Target organ

The liver and lung tumor incidence from chronic exposure biassays provide clear evidence of the carcinogenic potential of dichloromethane exposure. The biassays are supported by a substantial literature of genotoxicity and mechanistic studies (summarized in section 4.5). The evidence for mammary gland tumors from dichloromethane exposure is based primarily on observations of benign tumors in rats with inhalation exposure (NTP, 1986). Derivation of cancer potenciy values based on these data are presented in Appendix G. The potential brain cancer risk, suggested by a limited number of these relatively rare tumors in both animal and human studies, is identified as a data gap which would benefit from additional research.

Extrapolation approach

A route-to-route extrapolation from the NTP (1986) inhalation mouse bioassay was used to develop an oral cancer slope value. This value is inherently less certain than the values based on oral exposure due to the influence of route of exposure on toxicokinetics.

Dose Metric

There is considerable data supporting the role of GST-related metabolism of dichloromethane in carcinogenicity, as described in sections 4.5.1 and 4.7. Pretreatment of mice with buthionine sulphoximine, a GSH depletor, caused a decrease, to levels seen in controls, in the amount of DNA damage detected immediately after in vivo exposure in liver and lung tissue (Graves et al., 1995). Although the results of Landi et al. (2003) indicate that GST activity is not needed for the observation of DNA damage by the comet assay from some trihalomethanes (e.g., bromodichloromethane), the results for dichloromethane were much weaker and of uncertain significance.

Dose-response modeling

Because of the adequacy of the fit of the multistage model to the data, little modeling uncertainty would be expected to be introduced by the choice of this model. Application of the multistage model allowed for estimation of a point of departure in the lower region of exposure for observable cancer effects.

Low-dose extrapolation

The mode of action is a key consideration in determining how risks should be estimated for low-dose exposure. The in vitro and in vivo genotoxicity data suggest that mutagenicity is the most plausible mode of action, although key mutagenic events in the development of liver or lung tumors have not been identified. No data are available that provide an adequate rationale for choosing a nonlinear dose response in the low-dose region. Because a mutagenic mode of action is most plausible, a linear-low-dose extrapolation approach was used to estimate OSFs and IURs.

Interspecies extrapolation of dosimetry and risk

Target organ dosimetry for neoplastic mouse responses and estimation of equivalent internal human doses were accomplished using PBTK models for dichloromethane in mice and humans. Uncertainty in the ability of the PBTK models to estimate animal and human internal doses from lifetime bioassay low-level environmental exposures may affect the confidence in the cancer risk extrapolated from animal data. Uncertainties in the mouse and human model parameter values were integrated quantitatively into parameter estimation by utilizing hierarchical Bayesian methods to calibrate the models at the population level (David et al., 2006; Marino et al., 2006). The use of Monte Carlo sampling to define human model parameter distributions allowed for derivation of human distributions of dosimetry and cancer risk, providing for bounds on the recommended risk values.

A detailed discussion of PBTK model structure (CYP rate equation) and parameter uncertainties is provided in Section 5.3. While the structure and equations used in the existing model have been described in multiple peer-review publications over the past two decades, there are discrepancies between dichloromethane kinetics observed in vitro and the model parameters obtained from in vivo data. However, an alternative (dual-binding-site) CYP metabolic equation appears to resolve these discrepancies. Integration of the alternate rate equation into the PBTK modeling, and then quantitative risk assessment, will likely require several years of further research, and hence is beyond the scope of the current assessment. Since the GST activity in the current model is within a factor of three of that measured in vitro (when both are evaluated on a per gram of liver basis), the impact of that model uncertainty is also expected to be no more than a factor of three. Sensitivity to the human PBTK parameter distributions was evaluated by rescaling the parameters to the mean values obtained by David et al. (2006) for a specific data set (DiVincenzo and Kaplan, 1981) for which the GST activity was close to a numerical average of those obtained across individual data sets. When this was done, the upper bound estimates on GST dosimetry (for low, fixed inhalation or oral exposures) in the GST-T1 +/+ subpopulation increased by over an order of magnitude, as did the estimate of the mean activity for an inhalation exposure, although the estimated mean GST activity for an oral exposure only increased about two-fold. So while correspondence of the in vivo GST activity with that measured in vitro suggests a lower degree of quantitative uncertainty, it is possible that revision of the PBTK model could have a larger impact. The ultimate impact will depend on how revisions effect model predictions for both the animal and the human. If the predicted GST metabolism per unit exposure increases in both mice and humans by a similar factor, there will be little impact on the risk estimate. But if the GST activity predicted in the mouse is decreased by a factor of 3, while that in the human is increased by a factor of 3, for example, then the net impact would be an increase of 9-fold in human risk estimates.

The PBTK animal models were utilized deterministically; i.e., the single-value parameter estimates for the rat PBTK model were used for rat dosimetry simulations and the mean parameter estimates from the Bayesian analysis of Marino et al. (2006) were used for the mouse dosimetry simulations. To assess the effect of using point estimates of parameter values for calculation of rodent dosimetry, a sensitivity analysis was performed to identify model parameters most influential on the predictions of dose metrics used to estimate oral and inhalation cancer risks. As was described in the RfD and RfC sensitivity analysis calculation, this procedure used a univariate analysis in which the value of an individual model parameter was perturbed by an amount (Δ), in the forward and reverse direction (i.e., an increase and decrease from the nominal value), and the change in the output variable was determined. Results are for the effects of a perturbation of ±1% from the nominal value of each parameter on the output values at the end of a minimum of 10,000 simulated hours. This time was chosen to achieve a stable daily value of the dose metric, given that the simulated bioassay exposures did

not include weekend exposures. The exposure conditions represented the lowest bioassay exposure, resulting in significant increases in the critical effect. For inhalation exposures in mice, the blood:air partition coefficient, followed closely by the first-order GST-mediated metabolism rate (k_fC), had the greatest impact on the dose metric for liver cancer (mg dichloromethane metabolized via GST pathway per liter liver per day) (Figure 5-17). For drinking water exposures in mice, the k_fC, followed by the CYP-mediated maximum reaction velocity ($V_{max}C$), affected the liver cancer dose metric to the greatest extent (Figure 5-18). For mice inhaling dichloromethane, the lung cancer dose metric (mg dichloromethane metabolized via GST pathways per liter lung per day), like the liver cancer metric, was highly affected by the k_fC and the blood:air partition coefficient (Figure 5-19). However, since GST-mediated lung metabolism is calculated as a constant fraction of the liver metabolism rate ($A2 \times k_fC$), the lung cancer dose metric was most sensitive to the proportional yield of liver GST-mediated metabolic activity attributed to the lung. The blood:air partition coefficient was experimentally determined, lending high confidence to its value. Values for the three metabolic parameters were determined by computational optimization against data sets not directly measuring dichloromethane or its metabolites in the target/metabolizing tissues. It is uncertain how alternative values for these three parameters would affect the estimation of animal $BMDL_{10}$ values and, ultimately, the OSFs and IURs.

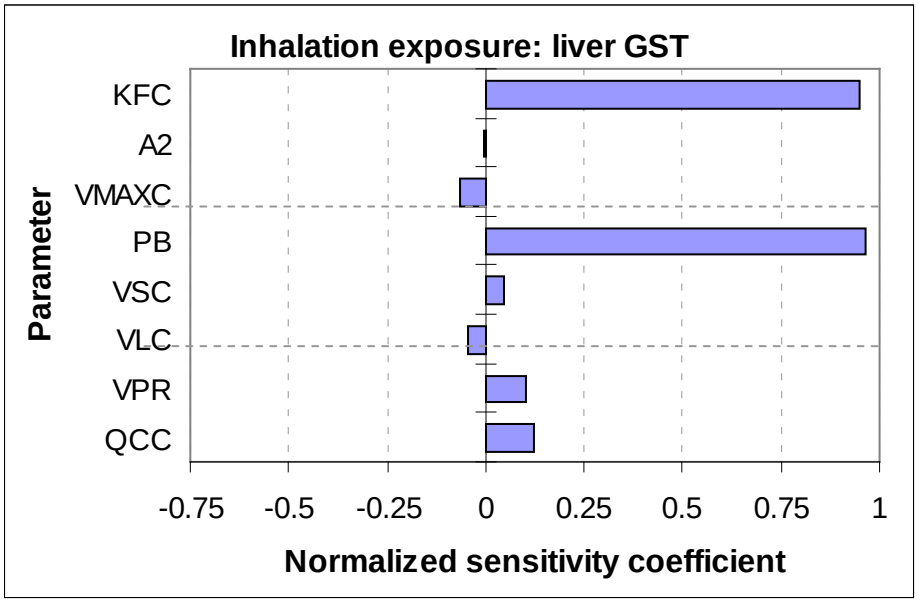

Figure 5-17. Sensitivity coefficients for long-term mass GST-mediated metabolites per liver volume from a long-term average daily inhalation concentration of 2000 ppm in mice. KFC = GST-mediated metabolism rate; A2 = proportion of liver GST metabolism attributed to the lung; VMAXC = CYP-mediated maximum rate of metabolism; PB = blood:air partition coefficient; VSC = slowly perfused tissue volume; VLC = liver volume; VPR = Ventilation perfusion ratio ; QCC = cardiac output constant.

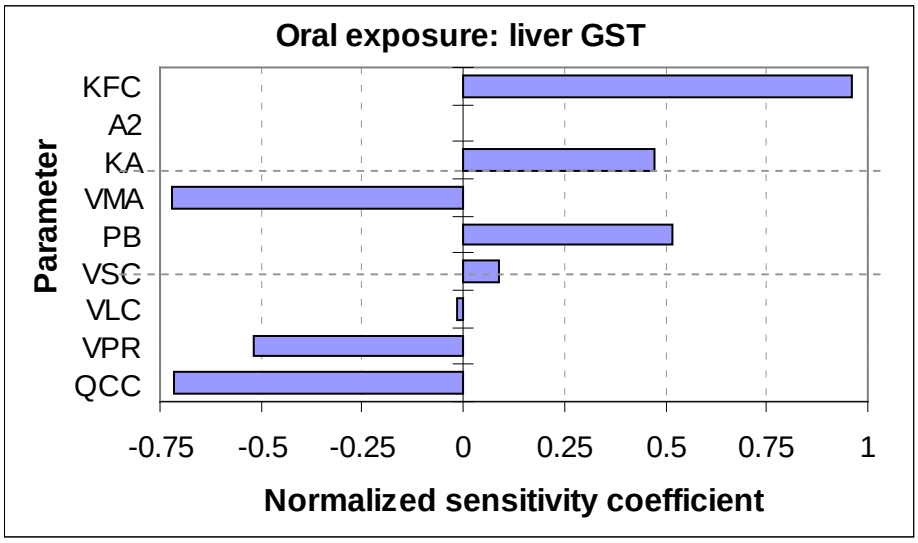

Figure 5-18. Sensitivity coefficients for long-term mass GST-mediated metabolites per liver volume from a long-term average daily drinking water concentration of 500 mg/L in mice. KFC = GST-mediated metabolism rate; A2 = proportion of liver GST metabolism attributed to the lung; KA = oral absorption rate from gut; VMAXC = CYP-mediated maximum rate of metabolism; PB = blood:air partition coefficient; VSC = slowly perfused tissue volume; VLC = liver volume; VPR = Ventilation perfusion ratio ; QCC = cardiac output constant.

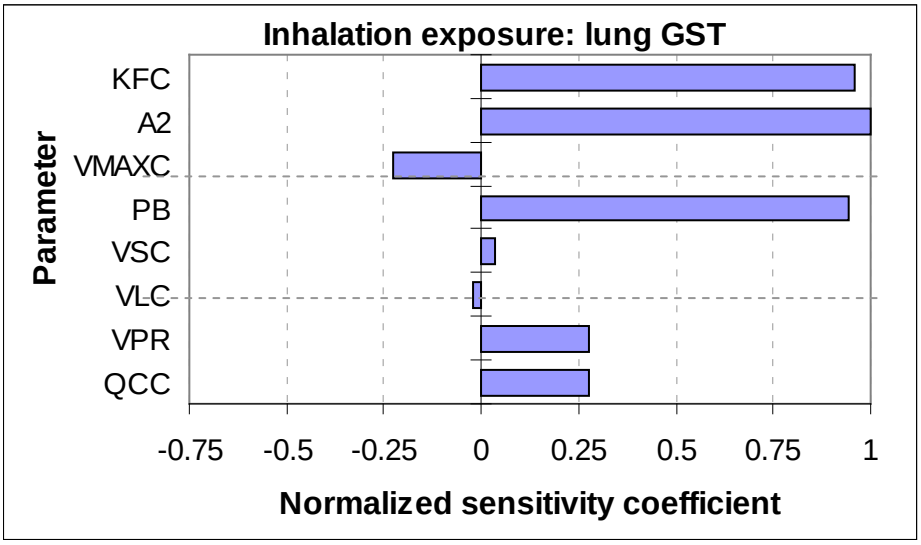

Figure 5-19. Sensitivity coefficients for long-term mass GST-mediated metabolites per lung volume from a long-term average daily inhalation concentration of 500 ppm in mice. KFC = GST-mediated metabolism rate; A2 = proportion of liver GST metabolism attributed to the lung; VMAXC = CYP-mediated maximum rate of metabolism; PB = blood:air partition coefficient; VSC = slowly perfused tissue volume; VLC = liver volume; VPR = Ventilation perfusion ratio ; QCC = cardiac output constant.

The comparison of the OSF derived from the oral exposure data and from the route–to-route extrapolation from the inhalation data provides a crude measure of the uncertainty in recommending a human OSF based on the available rodent bioassay data. The oral cancer slope factor based on route-to-route extrapolations from liver tumors in mice exposed by inhalation are about an order of magnitude lower than those based on the liver tumor responses in mice exposed via drinking water. This difference may be explained, at least partially, by the heterogeneity of hepatic cell types within the sinusoid, resulting in regio-specific toxicity. Oral exposure may result in a higher internal exposure of hepatocytes in the periportal region (particularly those lining the portal vein, through which all gastrointestinal-absorbed dichloromethane passes) than in the centrilobular region (Syracuse Research Corporation [SRC], 1989). Further, the metabolic capacity of hepatic cells is also regio-specific, with higher CYP activity found in the centrilobular region compared to the periportal region. Thus, liver perfusion via the systemic arterial circulation (through which inhaled dichloromethane would be introduced) or portal drainage of the gastrointestinal tract may influence regio-specific hepatotoxicity, resulting in the route-of-exposure effects on toxicity. The available PBTK models do not have the capability to predict regio-specific disposition of dichloromethane in the liver.

There is uncertainty as to whether the reactivity of the toxic dichloromethane metabolites is sufficiently high enough to preclude systemic distribution. Therefore, alternative derivations of cancer risk values were performed under the assumption that high reactivity leads to complete clearance from the tissue in which the active metabolite is formed (scaling factor = 1.0). The difference in scaling factor (7.0 for allometric scaling versus 1.0) results in a 7-fold decrease in estimated cancer toxicity values. Using a whole-body GST metabolism dose metric, the resulting OSF and IUR for liver and lung cancer were approximately five-fold higher than when tissue-specific dose metrics were used (Table 5-16 and Table 5-22). The mechanistic data support the notion that reactive metabolites produced in the target tissues are not well distributed and produce deleterious effects in the metabolizing tissues soon after generation. Thus, there is less uncertainty in the cancer risk values derived by using a tissue-specific GST metabolism dose metric compared with those derived using a whole-body GST metabolism dose metric.

Sensitive human populations

Possible sensitive populations include persons with altered CYP (e.g., obese individuals, alcoholics, diabetics, and the very young) and GST (e.g., GST-T1 homozygous conjugators) metabolic capacity. The PBTK model includes an estimate of the variability of CYP metabolism (sixfold variation), within the general population but does not specifically address what could be greater variation in these other groups. However, the known polymorphisms for GST-T1 expression were integrated into the human model. The distributions of human IUR values (from which the recommended [i.e., mean] values were taken) show that the 99th percentiles are

approximately 4-fold and 6-fold higher than means for liver and lung cancer. For the distribution of OSFs, the 99th percentile is approximately threefold higher than mean for liver cancer.

To further characterize the potential sensitivity of specific subpopulations, internal dose distributions for oral exposure to 1 mg/kg-day or inhalation exposure to 1 mg/m^3 were estimated for 1-year-old children and 70-year-old men and women to compare with the broader population results used to estimate cancer risks above. Since the recommended cancer risk estimate is based on the GST-T1$^{+/+}$ subpopulation, this analysis was also restricted to that subpopulation, so that only the factors of age and gender would differ. The impact of considering other GST-T1 groups can be seen where risk estimates for the GST-T1$^{+/-}$ and entire population mix are given above. Specification of age- and gender-specific parameters are as described in Appendix B. This sensitivity analysis is qualitatively similar to that described previously for the noncancer assessments of dichloromethane, where the variability in human equivalent administered dose and HEC values was estimated.

For this analysis, however, consideration of exclusively GST-T1$^{+/+}$ individuals will clearly narrow any estimate of variability. This analysis will also differ from that for noncancer effects in that the inverse of the former relationship is being considered (i.e., the variation in a specific internal dose for a fixed exposure is being computed, whereas for the human equivalent administered dose and HEC the variability in exposure levels corresponding to a fixed internal dose are estimated. The results of this analysis for oral exposures are shown in Figure 5-20 and Table 5-27 and for inhalation exposures in Figure 5-21 and Table 5-28.

For the oral exposure analysis, the distribution of internal doses shows little variation among the different age/gender groups (Figure 5-20, Table 5-27). The cancer analysis is based on a very low internal dose, where little enzymatic saturation is expected to occur, allowing for efficient first-pass metabolism, which is independent of differences in respiration; differences will be more significant at the higher doses analyzed for the noncancer human equivalent applied dose. Thus, the consideration of only GST metabolism and the narrower range of metabolic rate for that pathway in the +/+ population at low oral exposure rates results in minimal age/gender sensitivity differences (the 7-year-old female is only 5% more sensitive from pharmacokinetic factors than the general population).

For inhalation, an internal liver GST dose (mean value) about 2.5 times higher in the child than the general population is predicted due to the higher inhalation rate. The results for the liver GST dose for inhalation, Figure 5-21 and Table 5-28 indicate that the 70-year-old male and female populations are only slightly shifted from the general population, while the population for the 1-year-old child is a distinct upper tail of the general distribution.

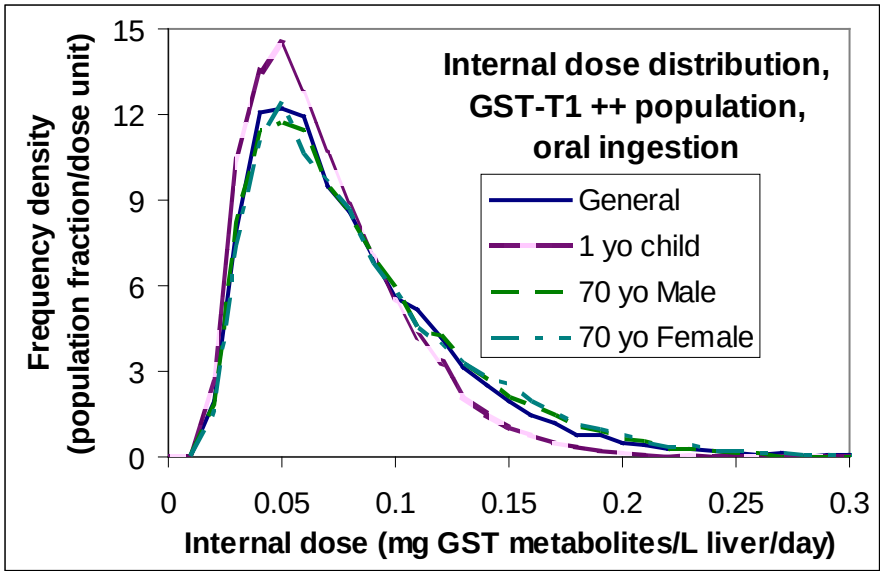

Figure 5-20. Histograms for a liver-specific dose of GST metabolism (mg GST metabolites per liter liver per day) for the general population (0.5- to 80-year-old males and females) and specific age/gender groups, within the population of GST-T1$^{+/+}$ genotypes, given a daily oral dose-rate of 1 mg/kg-day dichloromethane.

Table 5-27. Statistical characteristics of human internal doses for 1 mg/kg-day oral exposures in specific populations

Population	Internal dose (mg/L liver per day)[a]		
	Mean	95th percentile	99th percentile
All ages[b]	7.96×10^{-2}	1.91×10^{-1}	2.89×10^{-1}
1-year-old children	6.60×10^{-2}	1.47×10^{-1}	2.05×10^{-1}
70-year-old men	8.22×10^{-2}	1.97×10^{-1}	2.98×10^{-1}
70-year-old women	8.66×10^{-2}	2.18×10^{-1}	3.37×10^{-1}

[a]Liver-specific GST-T1 metabolism in GST-T1$^{+/+}$ individuals exposed orally to 1 mg/kg-day dichloromethane.
[b]0.5- to 80-year-old males and females.

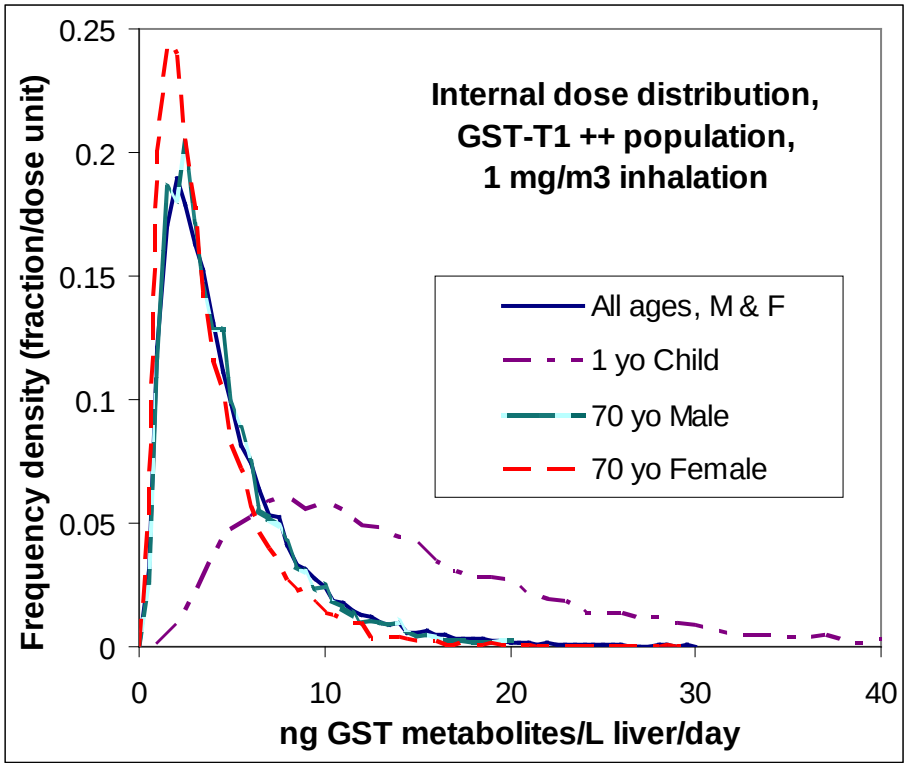

Figure 5-21. Histograms for liver-specific dose of GST metabolism (mg GST metabolites per liter liver per day) for the general population (0.5- to 80-year-old males and females) and specific age/gender groups, within the population of GST-T1$^{+/+}$ genotypes, given a continuous inhalation exposure to 1 mg/m^3 dichloromethane.

Table 5-28. Statistical characteristics of human internal doses for 1 mg/m^3 inhalation exposures in specific subpopulations

Population	Internal dose (mg/L liver per day)[a]		
	Mean	95th percentile	99th percentile
All ages[b]	5.63×10^{-6}	1.56×10^{-5}	$2.60s \times 10^{-5}$
1-year-old children	1.41×10^{-5}	3.30×10^{-5}	4.70×10^{-5}
70-year-old men	4.36×10^{-6}	1.11×10^{-5}	1.62×10^{-5}
70-year-old women	3.55×10^{-6}	9.41×10^{-6}	1.45×10^{-5}

[a]Liver-specific GST-T1 metabolism in GST-T1$^{+/+}$ individuals exposed continuously by inhalation to 1 mg/m^3 dichloromethane.
[b]0.5- to 80-year-old males and females.

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

6.1. HUMAN HAZARD POTENTIAL

Dichloromethane (CASRN 75-09-2), also known as methylene chloride, is a colorless liquid with a penetrating ether-like odor. It is produced by the direct reaction of methane with chlorine at either high temperatures or low temperatures under catalytic or photolytic conditions. The principal uses for dichloromethane have been in paint strippers and removers, as a propellant in aerosols, in the manufacture of drugs, pharmaceuticals, film coatings, electronics, and polyurethane foam, and as a metal-cleaning solvent.

Dichloromethane is rapidly absorbed through both oral administration and inhalation exposure with a steady-state saturation occurring with inhalation. Results from studies of animals show that, following absorption, dichloromethane is rapidly distributed throughout the body and has been detected in all tissues that have been evaluated. Metabolism of dichloromethane involves two primary pathways. Dichloromethane is metabolized to CO and to a lesser extent CO_2 in a CYP-dependent oxidative pathway (CYP2E1) that is predominant at low exposure levels. The other major pathway for dichloromethane metabolism involves the conjugation of dichloromethane to GSH, catalyzed by GST (GST-T1). This results in the formation of a GSH conjugate that is eventually metabolized to CO_2. The conjugation of dichloromethane to GSH results in the formation of two reactive intermediates that have been hypothesized to be involved in dichloromethane carcinogenicity, S-(chloromethyl)glutathione and formaldehyde. Formation of formaldehyde leads to several covalent modifications of cellular macromolecules, including DNA-protein cross-links (Casanova et al., 1996) and RNA-formaldehyde adducts (Casanova et al., 1997). Evidence is also available that S-(chloromethyl)glutathione can result in both DNA SSBs and DNA mutations, presumably through DNA alkylation (Green, 1997; Graves and Green, 1996; Graves et al., 1996, 1994a; Hashmi et al., 1994). However, DNA reaction products (e.g., DNA adducts) produced by S-(chloromethyl)glutathione have not been found, possibly due to potential instability of these compounds (Watanabe et al., 2004; Hashmi et al., 1994).

Information on noncancer effects in humans exposed orally to dichloromethane are restricted to case reports of neurological impairment (general CNS depression), liver and kidney effects (as severe as organ failure), and gastrointestinal irritation in individuals who ingested amounts ranging from about 25 to 300 mL (Chang et al., 1999; Hughes and Tracey, 1993). The animal toxicity database identifies hepatic effects (hepatic vacuolation, nonneoplastic liver foci) as the critical dose-dependent noncancer endpoint associated with oral exposure to dichloromethane. The most frequently observed liver effect was hepatocyte vacuolation, seen with drinking water exposure (90 days) in F344 rats at ≥166 mg/kg-day and $B6C3F_1$ mice at

586 mg/kg-day (Kirschman et al., 1986) and with gavage exposure (14 days) in CD-1 mice at 333 mg/kg-day (Condie et al., 1983). Hepatocyte degeneration or necrosis was observed in female F344 rats exposed in drinking water for 90 days to 1,469 mg/kg-day (Kirschman et al., 1986) and in female F344 rats exposed by gavage for 14 days to 337 mg/kg-day (Berman et al., 1995). In the chronic-duration (104-week) study, liver effects were described as nonneoplastic foci in F344 rats exposed to drinking water doses between 50 and 250 mg/kg-day (Serota et al., 1986a). In the reproductive oral administration studies, no significant effect on reproductive function or parameter was observed in rats up to 225 mg/kg-day (General Electric Co., 1976) or in mice up to 500 mg/kg-day (Raje et al., 1988). The NOAEL and LOAEL for altered neurological functions in female F344 rats were 101 and 337 mg/kg-day (as reported by Moser et al., 1995).

Acute inhalation exposure of humans to dichloromethane has been associated with cardiovascular impairments due to decreased oxygen availability from COHb formation and neurological impairment from interaction of dichloromethane with nervous system membranes (Bos et al., 2006; ACGIH, 2001; ATSDR, 2000; Cherry et al., 1983; Putz et al., 1979; Gamberale et al., 1975; Winneke, 1974). Relatively little is known about the long-term neurological effects of chronic exposures, although there are studies that provide some evidence of an increased prevalence of neurological symptoms among workers with average exposures of 75–100 ppm (Cherry et al., 1981) and long-term effects on some neurological measures (i.e., possible detriments in attention and reaction time in complex tasks) in workers whose past exposures were in the 100–200 ppm range (Lash et al., 1991). These studies are limited by the relatively small sample sizes and low power for detecting statistically significant results for these endpoints.

Following repeated inhalation to dichloromethane, the liver is the most sensitive target for noncancer toxicity in rats and mice. Lifetime exposure was associated with hepatocyte vacuolation and necrosis in F344 rats exposed to 1,000 ppm 6 hours/day (Mennear et al., 1988; NTP, 1986), hepatocyte vacuolation in Sprague-Dawley rats exposed to 500 ppm 6 hours/day (Nitschke et al., 1988a; Burek et al., 1984), and hepatocyte degeneration in $B6C3F_1$ mice exposed to 2,000 ppm 6 hours/day (lower concentrations were not tested in mice) (Mennear et al., 1988; NTP, 1986). Other effects observed include renal tubular degenerations in F344 rats and $B6C3F_1$ mice at 2,000 ppm, testicular atrophy in $B6C3F_1$ mice at 4,000 ppm, and ovarian atrophy in $B6C3F_1$ mice at 2,000 ppm.

Other studies with inhalation exposure to dichloromethane revealed no significant effects on reproductive performance in rats (up to 1,500 ppm) (Nitschke et al., 1988b), although some evidence of a decrease in fertility index was seen in male mice exposed to 150 and 200 ppm (Raje et al., 1988), and no adverse effects on fetal development of mice or rats exposed up to 1,250 ppm were seen by Schwetz et al. (1975). Decreases in fetal BW and changes in behavioral habituation were observed in Long-Evans rats exposed to 4,500 ppm during the gestational

period (Bornschein et al., 1980; Hardin and Manson, 1980). Exposure-related noncancer effects on the lungs consisted of foreign-body pneumonia in rats exposed to 8,400 ppm 6 hours/day for 13 weeks (NTP, 1986), Clara cell vacuolation in mice exposed to 4,000 ppm 6 hours/day for 13 weeks (Foster et al., 1992), and pulmonary congestion in guinea pigs exposed to 5,000 ppm 7 hours/day for 6 months (Heppel et al., 1944). Several neurological mediated parameters including decreased activity (Kjellstrand et al., 1985; Weinstein et al., 1972; Heppel and Neal, 1944), impairment of learning and memory (Alexeef and Kilgore, 1983), and changes in responses to sensory stimuli (Rebert et al., 1989) are reported from acute and short-term dichloromethane exposure. Evidence of a localized immunosuppressive effect in the lung, resulting from inhalation dichloromethane exposure, was seen in an acute exposure (3 hours, 100 ppm) study in CD-1 mice (Aranyi et al., 1986).

Numerous in vitro studies have demonstrated mutagenic and genotoxic effects associated with dichloromethane exposure. For example, bacterial assays, yeast, and fungi provide evidence that the mutagenic action of dichloromethane in bacterial systems is enhanced by metabolic activation (e.g., Dillon et al., 1992; Jongen et al., 1982; Gocke et al., 1981). Positive results from assays of DNA damage with in vitro mammalian systems provide support that dichloromethane genotoxicity is linked to metabolism by GST enzymes (Graves et al., 1996, 1995, 1994b). Consistent evidence for several genotoxic endpoints in target tissues (liver and lung) in mice following in vivo exposure to dichloromethane provides supporting evidence that GST-pathway metabolites are key actors in the mutagenic and carcinogenic mode of action for dichloromethane. Pretreatment of mice with buthionine sulphoximine, a GSH depletor, caused a decrease to levels seen in controls in the amount of DNA damage detected immediately after in vivo exposure in liver and lung tissue, indicating GSH involvement in the genotoxic process (Graves et al., 1995). DNA damage (detected by the comet assay) was also reported in liver and lung tissues from male CD-1 mice sacrificed 4 hours after administration of a single oral dose of 1,720 mg/kg of dichloromethane (Sasaki et al., 1998). In this study, DNA damage in lung and liver was not detected 3 hours after dose administration, and no DNA damage occurred at either time point in several other tissues in which a carcinogenic response was not seen in chronic animal cancer bioassays (e.g., stomach, kidney, bone marrow). The weight of evidence from these studies suggests that dichloromethane is carcinogenic by a mutagenic mode of action.

Dichloromethane is "likely to be carcinogenic in humans" by the inhalation and oral routes of exposure under the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a). Results from several 2-year bioassays provide adequate evidence of the carcinogenicity of dichloromethane in mice and rats exposed by inhalation, as well as adequate data to describe dose-response relationships. Oral exposure to dichloromethane produced statistically significant increases in hepatocellular adenomas and carcinomas in male $B6C3F_1$ mice (Serota et al., 1986b; Hazelton Laboratories, 1983). Inhalation exposure to concentrations of 2,000 or 4,000 ppm dichloromethane produced increased incidences of lung and liver tumors in $B6C3F_1$ mice

(Maronpot et al., 1995; Foley et al., 1993; Kari et al., 1993; Mennear et al., 1988; NTP, 1986). Significantly increased incidences of benign mammary tumors (adenomas or fibroadenomas) were observed in male and female F344/N rats exposed by inhalation to 2,000 or 4,000 ppm (Mennear et al., 1988; NTP, 1986). A statistically significant increased incidence of brain or CNS tumors has not been observed in any of the animal cancer bioassays, but a 2-year study using relatively low exposure levels (0, 50, 200, and 500 ppm) in Sprague-Dawley rats observed a total of six astrocytoma or glioma (mixed glial cell) tumors in the exposed groups (Nitschke et al., 1988a). These tumors are exceedingly rare in rats, and there are few examples of statistically significant trends in animal bioassays (Sills et al., 1999).

6.2. DOSE-RESPONSE

6.2.1. Oral RfD

The available oral toxicity data for animals identify hepatic effects (hepatic vacuolation, nonneoplastic liver foci) as the most sensitive noncancer endpoint associated with chronic oral exposure to dichloromethane. The 104-week drinking-water study in F344 rats (Serota et al., 1986a) was selected as the principal study for RfD derivation because the study provided a sensitive endpoint (nonneoplastic liver foci) and used lower doses in comparison to other chronic oral administration studies. In this study, four doses (6, 52, 125, and 235 mg/kg-day in males; 6, 58, 136, and 263 mg/kg-day in females) were used. A NOAEL of 6 mg/kg-day in males and females and a LOAEL of 52 (male) and 58 (female) mg/kg-day for nonneoplastic alterations of liver foci was identified.

An RfD of 7×10^{-3} mg/kg-day is recommended for use in humans. The RfD derivation process involved first fitting all available dichotomous models in BMDS version 2.0 to the incidence data for male rats. The male data were used because a greater sensitivity was seen in males compared with females in this study. A dose metric of average daily mass of dichloromethane metabolized via the CYP pathway per unit volume of liver was derived from an EPA-modified rat PBTK model (see Appendix C). This metric was chosen because there are no data to support the role of a specific metabolite in the development of the noncancer liver lesions seen in oral and inhalation exposure studies and the CYP-metabolism dose metric was determined to be most consistent with the data. Then, the lower 95% confidence limit on the dose associated with a 10% risk for liver lesions ($BMDL_{10}$) was derived, based on the best fitting model (in terms of the value of the AIC and examination of model fit and residuals). Because the metric is a rate of metabolism, rather than the concentration of putative toxic metabolites, and the clearance of these metabolites may be slower per volume tissue in the human compared with the rat, this rodent internal dose metric for noncancer effects was adjusted by dividing by a pharmacokinetic scaling factor to obtain a human equivalent internal $BMDL_{10}$. This $BMDL_{10}$ was then converted to the human equivalent dose by using a human PBTK model (adapted from David et al., 2006; see Appendix B) that utilizes Monte Carlo sampling techniques to provide a

distribution of human equivalent doses. The first percentile of the distribution of human equivalent doses was chosen to include the most sensitive population, while staying within bounds of what is considered computationally stable. The first percentile human equivalent administered dose was used as a point of departure and was divided by a composite UF of 30 (3 [$10^{0.5}$] to account for uncertainty about interspecies toxicodynamic equivalence, 3 [$10^{0.5}$] to account for uncertainty about toxicodynamic variability in humans, and 3 [$10^{0.5}$] for database deficiencies) to arrive at an RfD of 7×10^{-3} mg/kg-day.

Use of the mean value (3.6×10^{-1} mg/kg-day) of the human equivalent administered dose distribution instead of the 1st percentile, with an additional UF of 3 ($10^{0.5}$) to account for human toxicokinetic variability, would yield a candidate RfD of 4×10^{-3}, which is relatively similar to the recommended RfD of 1×10^{-3}.

6.2.2. Inhalation RfC

The liver is the most sensitive target for noncancer toxicity in rats and mice, following repeated inhalation exposure to dichloromethane. Nonneoplastic liver lesions (specifically, hepatic vacuolation) in rats are the critical noncancer effect from chronic dichloromethane inhalation in animals. Inhalation bioassays with Sprague-Dawley rats identified the lowest inhalation LOAEL for nonneoplastic liver lesions in the database: 500 ppm (6 hours/day, 5 days/week for 2 years) (Nitschke et al., 1988a; Burek et al., 1984). Nitscke et al. (1988a) identified a NOAEL of 200 ppm for hepatocyte vacuolation in female rats. Because the Nitschke et al. (1988a) study more adequately covers the range spanning the BMR compared with the study by Burek et al. (1984), the former study was selected as the principal study for derivation of a chronic inhalation RfC.

An RfC of 0.2 mg/m^3 is derived based on the observed critical effect in the principal study. As was described above for the RfD, the RfC derivation process was based on a dose metric of average daily mass of dichloromethane metabolized via the CYP pathway per unit volume of liver. This metric was derived from an EPA-modified rat PBTK model (see Appendix C). Then, the lower 95% confidence limit on the dose associated with a 10% risk for liver lesions ($BMDL_{10}$) was derived, based on the best fitting model in terms of the value of the AIC and examination of model fit and residuals. Because the metric is a rate of metabolism, rather than the concentration of putative toxic metabolites, and the clearance of these metabolites may be slower per volume tissue in the human compared with the rat, this rodent internal dose metric for noncancer effects was adjusted by dividing by a pharmacokinetic scaling factor to obtain a human-equivalent internal $BMDL_{10}$. This $BMDL_{10}$ was then converted to the HEC by using a human PBTK model (adapted from David et al., 2006; see Appendix B) that utilizes Monte Carlo sampling techniques to provide a distribution of HECs.

The first percentile HEC was used as a point of departure. This percentile was chosen because it included the most sensitive population while staying within bounds of what is

considered computationally stable. This point of departure was divided by a composite UF of 100 (3 [$10^{0.5}$] to account for uncertainty about interspecies toxicodynamic equivalence, 3 [$10^{0.5}$] to account for uncertainty about toxicodynamic variability in humans, and 10 for database deficiencies) to arrive at an RfC of 0.2 mg/m^3.

Use of the mean value (47.36 mg/m^3)of the HEC distribution instead of the 1st percentile, with an additional UF of 3 ($10^{0.5}$) to account for human toxicokinetic variability would yield a candidate RfC identical to the recommended value of 0.2 mg/m^3. In addition, two comparison values derived from occupational studies produced values of 3.5 mg/m^3 (Cherry et al., 1983) and 0.55 mg/m^3 (Lash et al., 1991). The animal-derived candidate RfC is preferable to the human-derived candidate RfC because of the uncertainties about the exposure durations for the workers in the Cherry et al. (1983) study and uncertainties regarding the exposures and effect sizes in Lash et al. (1991) and because the RfC based on the rat data is more health protective.

6.2.3. Uncertainties in Reference Dose and Reference Concentration Values

One data uncertainty identified is the potential for neurodevelopmental effects. Animal bioassays have not identified gross or microscopic effects on neural tissues from long-term exposures or single (Schwetz et al., 1975) or multigenerational (Nitschke et al., 1988b) developmental toxicity studies. However, behavioral changes were observed in pups born to rats exposed to high levels (4,500 ppm) of dichloromethane (Bornschein et al., 1980; Hardin and Manson, 1980); 4,500 ppm was the only dose used in this study. Thus uncertainty exists as to the development of neurological effects from lower gestational exposures in animals, or in humans. In addition, immunotoxicity data revealed an additional area of data uncertainty specifically with respect to inhalation exposure. Data from Aranyi et al. (1986) demonstrated evidence of immunosuppression, following a single 100 ppm dichloromethane exposure for 3 hours in CD-1 mice. The weight of evidence for nonneoplastic effects in humans and animals suggests that the development of liver lesions is the most sensitive effect, with a UF applied because of the lack of neurodevelopmental studies and, for the RfC, the uncertainty regardingthe lack of a low dose developmental study.

The extrapolation of internal dichloromethane dosimetry from nonneoplastic rat responses to human risk was accomplished by using PBTK models for dichloromethane in rats and humans. Uncertainties in rat and human dosimetry used for RfD and RfC derivation can arise from uncertainties in the PBTK models to accurately simulate the toxicokinetics of dichloromethane for animals under bioassay conditions and humans experiencing relatively low, chronic environmental exposures. Further, the dose metric used in the models is the rate of metabolism to a putative toxic metabolite, rather than the concentration (average or area under the concentration curve of the metabolite), so the model specifically fails to account for rodent-human differences in clearance or removal of the toxic metabolite. A scaling factor, based on BW ratios, was used to account for this difference.

Uncertainties in the human population model were quantitatively accounted for by utilizing hierarchical Bayesian calibration methods during model development (David et al., 2006; Marino et al., 2006). The rat model was modified and utilized in a deterministic manner. Data were not available to perform a hierarchical Bayesian calibration in the rat, but uncertainties in the rat model predictions were assessed qualitatively. For both oral and inhalation exposures, the liver volume, followed closely by the volume of slowly perfused tissues, had the greatest impact on the internal dose of mg dichloromethane metabolized via CYP pathway per liter tissue per day. This was due to the fact that the dose metric is a tissue-specific concentration, the majority of CYP metabolism is attributed to the liver, and changes in liver volume have a greater impact on the total CYP metabolism than either of the individual V_{max} values. There is high confidence in the values used for volume of liver and slowly perfused tissues in the rat, as these are well studied (Brown et al., 1997). Therefore, the uncertainties associated with use of the rat PBTK model should not markedly affect the values of the RfD and RfC.

An additional uncertainty inherent in this process, however, is the lack of knowledge concerning the most relevant dose metric (e.g., a specific metabolite) within the context of the development of the noncancer liver effects. This basic research question represents a data gap, and this uncertainty is not addressed quantitatively or qualitatively in the assessment.

The effect of dichloromethane on human populations that are sensitive due to pharmacokinetic differences was addressed quantitatively by using a human probabilistic PBTK model to generate distributions of human exposures likely to occur given a specified internal $BMDL_{10}$. The model and resulting distributions take into account the known differences in human physiology and metabolic capability with regard to dichloromethane dosimetry. The first percentile values of the distributions of human equivalent doses (Table 5-3) and HECs (Table 5-7) served as points of departure for candidate RfDs and RfCs, respectively, to protect toxicokinetically sensitive individuals. No data are available regarding toxicodynamic differences within a human population. Therefore, a UF of 3 for possible differences in human toxicodynamic responses is intended to be protective for sensitive individuals.

6.2.4. Oral Cancer Slope Factor

The recommended oral cancer slope factor for dichloromethane is 1×10^{-3} $(mg/kg\text{-}day)^{-1}$, which is based on liver tumor responses in male $B6C3F_1$ mice exposed to dichloromethane in drinking water for 2 years (Serota et al., 1986b; Hazelton Laboratories, 1983). This value was derived by using a tissue-specific GST metabolism dose metric with allometric scaling to account for uncertainty regarding the reactivity and clearance of the metabolite(s) involved in the carcinogenic response.

There was only one adequate oral exposure cancer bioassay (Serota et al., 1986a, b; Hazelton Laboratories, 1983) evaluating the carcinogenic potential of orally administered

dichloromethane in F344 rats and $B6C3F_1$ mice. Significant increases in incidence of liver adenomas and carcinomas were observed in male (trend *p*-value = 0.058) but not female $B6C3F_1$ mice (Serota et al., 1986b; Hazelton Laboratories, 1983). In F344 rats (Serota et al., 1986a), no increased incidence of liver tumors was seen in male rats, and the pattern in female rats was characterized by a jagged stepped pattern of increasing incidence of hepatocellular carcinoma or neoplastic nodules; a similar pattern, but based on more sparse data, was seen when limited to hepatocellular carcinomas. Statistically significant increases in tumor incidences were observed in the 50 and 250 mg/kg-day groups (incidence rates of 10 and 14%, respectively) but not in the 125 mg/kg-day group (incidence rate of 3%). Incidence was also increased (10%) in a group exposed for 78 weeks followed by a 26-week period of no exposure. The derivation of the oral cancer slope factor is based on the male mice data because of their greater sensitivity to liver cancer compared with female rats.

A modified mouse PBTK model of Marino et al. (2006) was used to approximate the internal dose of daily dichloromethane (mg) metabolized via the GST pathway per unit volume of liver from the daily oral administered doses. This approach was taken based on evidence that GST-pathway metabolites produced from dichloromethane are primarily responsible for dichloromethane carcinogenicity in mouse liver. The multistage dose-response model (BMDS version 2.0) was used to fit the mouse liver tumor incidence and PBTK model-derived internal dose data and derive a mouse internal BMD and BMDL associated with 10% extra risk ($BMDL_{10}$). The human $BMDL_{10}$ was derived by multiplying the mouse $BMDL_{10}$ by allometric scaling factor $(BW_{human}/BW_{mouse})^{0.25} \approx 7$). Linear extrapolation from the internal human $BMDL_{10}$ ($0.1/BMDL_{10}$) was used to derive oral risk factors for liver tumors based on tumor responses in male mice. The linear low-dose extrapolation approach for agents with a mutagenic mode of action was selected because GST-metabolism of dichloromethane is expected to occur at and below exposures producing the mouse $BMDL_{10}$, even though CYP2E1 metabolism is expected to be unsaturated and to represent the predominant metabolic pathway in the liver. Currently, there are no data from chronic oral cancer bioassays in mice providing support for a nonlinear dose-response relationship.

Probability distributions of human oral cancer slope factors were derived by using a human PBTK model (adapted from David et al. [2006]; see Appendix B). The cancer reference values (OSF and IUR) were derived for a sensitive population: a population composed entirely of carriers of the GST-$T1^{+/+}$ homozygous genotype (that is, the group that would be expected to be most sensitive to the carcinogenic effects of dichloromethane). In addition, cancer values derived for a population reflecting the estimated frequency of GST-T1 genotypes in the current U.S. population (20% GST-$T1^{-/-}$, 48% GST-$T1^{+/-}$, and 32% GST-$T1^{+/+}$) were presented. All simulations also included a distribution of CYP activity based on data from Lipscomb et al. (2003). The mean OSF based on liver tumors in mice exposed to dichloromethane in drinking water, 1×10^{-3} $(mg/kg\text{-}day)^{-1}$, based on what is assumed to be the most sensitive of the

populations, the GST-T1$^{+/+}$ group, is the recommended OSF to be used in deterministic risk assessments for chronic oral exposures to dichloromethane.

An OSF derived from the liver tumor data in the Serota et al. (1986b) study, using administered dose dosimetry rather than PBTK modeling, is approximately one order of magnitude higher than the current recommended value of 1×10^{-3} (per mg/kg-day). There is approximately one to two orders of magnitude difference among the values based on different dose metrics, scaling factors, and populations (Table 6-1).

The recommended OSF of 1×10^{-3} (per mg/kg-day) is based on a tissue-specific GST internal dose metric with allometric scaling. Although the involvement of the GST pathway in carcinogenic response has been established, some uncertainty remains as to the metabolite(s) involved and the rate at which those metabolites are cleared. The value derived specifically for the GST-T1$^{+/+}$ population is recommended to provide protection for the population that is hypothesized to be most sensitive to the carcinogenic effect. Application of ADAFs to the cancer OSF is recommended in combination with appropriate exposure data when assessing risks associated with early-life exposure (see section 5.4.4 for more details).

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Table 6-1. Comparison of OSFs derived by using various assumptions and metrics, based on liver tumors in male mice

Population[a]	Dose metric	Species, sex	Tumor	Scaling factor	Mean OSF $(mg/kg\text{-}day)^{-1}$	Source (Table)
GST-T1$^{+/+}$	**Tissue-specific GST-metabolism rate[b]**	**Mouse, male**	**Liver**	**7.0**	**1.4×10^{-3}**	**Table 5-13**
	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	2.0×10^{-4}	Table 5-13
	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	8.1×10^{-4}	Table 5-13
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	7.0	1.0×10^{-4}	Table 5-14
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	1.0	1.5×10^{-5}	Table 5-14
	Route-to-route extrapolation, whole-body metabolism	Mouse, male	Liver	7.0	5.8×10^{-5}	Table 5-14
Mixed	Tissue-specific GST-metabolism rate[b]	Mouse, male	Liver	7.0	8.0×10^{-4}	Table 5-13
	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	1.2×10^{-4}	Table 5-13
	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	4.6×10^{-4}	Table 5-13
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	7.0	5.8×10^{-5}	Table 5-14
	Route-to-route extrapolation, tissue-specific metabolism	Mouse, male	Liver	1.0	8.3×10^{-6}	Table 5-14
	Route-to-route extrapolation, whole-body metabolism	Mouse, male	Liver	7.0	3.3×10^{-5}	Table 5-14
	Applied dose (human equivalent dose)	Mouse, male	Liver		1.0×10^{-2}	Table 5-15
	1995 IRIS assessment	Mouse, male	Liver		7.5×10^{-3}	

[a]GST-T1$^{+/+}$ = homozygous, full enzyme activity; Mixed = genotypes based on a population reflecting the estimated frequency of genotypes in the current U.S. population: 20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).

Bolded value is the basis for the recommended OSF of 1×10^{-3} per mg/kg-day.

6.2.5. Cancer Inhalation Unit Risk

The recommended cancer IUR is 1×10^{-8} $(\mu g/m^3)^{-1}$ for the development of liver and lung cancers, based on data from male $B6C3F_1$ mice, using a tissue-specific GST metabolism dose metric. Data for liver and lung tumors in male and female $B6C3F_1$ mice, following exposure to airborne dichloromethane, were used to develop IURs for dichloromethane (Mennear et al., 1988; NTP, 1986). This study was selected as the principal study to derive an IUR for dichloromethane because of the completeness of the data, adequate sample size, and clear dose response. In the NTP (1986) study, significant increases in incidence of liver and lung adenomas and carcinomas were observed in both sexes of $B6C3F_1$ mice exposed 6 hours/day, 5 days/week for 2 years.

The PBTK model of Marino et al. (2006) for dichloromethane in the mouse was used to calculate long-term daily average internal liver doses. The selected internal dose metrics for liver tumors and lung tumors were long-term average daily mass of dichloromethane metabolized via the GST pathway per unit volume of liver and lung, respectively. This approach was taken based on evidence that GST-pathway metabolites produced from dichloromethane are primarily responsible for dichloromethane carcinogenicity in mouse liver. The multistage dose-response model (BMDS version 2.0) was used to fit the mouse liver tumor incidence and PBTK model-derived internal dose data and derive a mouse internal BMD and BMDL associated with 10% extra risk ($BMDL_{10}$). The human $BMDL_{10}$ was derived by multiplying the mouse $BMDL_{10}$ by allometric scaling factor $(BW_{human}/BW_{mouse})^{0.25} \approx 7$). A linear extrapolation approach using the internal human $BMDL_{10}$ for liver and lung tumors was used to calculate human tumor risk factors by dividing the BMR of 0.1 by the human BMDL for each tumor type. Currently, there are no data from chronic inhalation cancer bioassays in mice or rats providing support for a nonlinear dose-response relationship.

The human PBTK model (adapted from David et al. [2006]; see Appendix B) provided distributions of human internal dose metrics of daily mass of dichloromethane metabolized via the GST pathway per unit volume of liver and lung resulting from chronic inhalation exposure to a unit concentration of 1 $\mu g/m^3$ dichloromethane (0.00029 ppm). As with the OSF, the cancer IUR was derived for a sensitive population: a population composed entirely of carriers of the GST-T1 homozygous positive genotype (that is, the group that would be expected to be most sensitive to the carcinogenic effects of dichloromethane). In addition, cancer values derived for a population reflecting the estimated frequency of GST-T1 genotypes in the current U.S. population (20% GST-$T1^{-/-}$, 48% GST-$T1^{+/-}$, and 32% GST-$T1^{+/+}$) were also presented. The distributions of IURs for liver or lung tumors were generated by multiplying the human tumor risk factor for each tumor type and sex by the distribution of internal doses from chronic exposure to 1 $\mu g/m^3$ dichloromethane. A procedure to combine risks for liver and lung tumors using different dose metrics for the different tumors (i.e., liver-specific and lung-specific

metabolism for liver and lung tumors, respectively), was used to derive the recommended IUR of 1×10^{-8} $(\mu g/m^3)^{-1}$ based on what is assumed to be the most sensitive of the populations, the GST-T1$^{+/+}$ group.

The current recommended IUR value of 1×10^{-8} $(\mu g/m^3)^{-1}$ is approximately 1.5 orders of magnitude lower than the previous IRIS value of 4.7×10^{-7} $(\mu g/m^3)^{-1}$ and similar to the occupational exposure-based risk value of 4.17×10^{-8} $(\mu g/m^3)^{-1}$ promulgated by OSHA (1997) and derived from an estimated risk of 3.62×10^{-3} for a lifetime occupational inhalation exposure of 25 ppm. The current use of the updated mouse PBTK model, with blood and tissue equilibrium partition coefficients and metabolic parameters updated with MCMC calibration, resulted in approximately three- and four-fold increases in the values of internal liver and lung dose metrics, respectively, associated with the dichloromethane exposure concentrations, compared with estimates from the model used in the previous IRIS assessment. For a unit inhalation exposure, the mean internal *lung GST* dose predicted for the entire population predicted by the MCMC updated human PBTK model is approximately thirteen-fold lower compared with the human PBTK model used in the U.S. EPA (1995) assessment. The mean internal *liver GST* dose, however, is approximately the same as (only 16% higher than) that obtained with the previous PBTK parameters. For unit oral exposures, the mean internal *liver GST* dose predicted by the MCM updated model is about 80% of that predicted using the previous parameters, while the mean *whole-body GST* dose is predicted to be about 50% of that predicted using the previous parameters.

An IUR derived from the liver tumor data of the NTP (1986) study using applied concentration dosimetry rather than PBTK modeling, 3.7×10^{-7} $(\mu g/m^3)^{-1}$, is approximately one order of magnitude higher than the currently recommended value of 1×10^{-8} $(\mu g/m^3)^{-1}$ (Table 6-2). There is approximately one- to two- orders of magnitude difference among the values based on different dose metrics, scaling factors, and populations.

The recommended IUR value of 1×10^{-8} $(\mu g/m^3)^{-1}$ is based on a tissue-specific GST-internal dose metric with allometric scaling. Although the involvement of the GST pathway in carcinogenic response has been established, some uncertainty remains as to the metabolite(s) involved and the rate at which those metabolites are cleared. The value derived specifically for the GST-T1$^{+/+}$ population is recommended to provide protection for the population that is hypothesized to be most sensitive to the carcinogenic effect. Application of ADAFs to the cancer IUR is recommended when assessing risks associated with early-life exposure (see section 5.4.4 for more details).

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Table 6-2. Comparison of IURs derived by using various assumptions and metrics

Population[a]	Dose metric	Species, sex,	Tumor type	Scaling factor	IUR[b] $(\mu g/m^3)^{-1}$	Source (Table)
GST-T1$^{+/+}$	**Tissue-specific GST-metabolism rate[c]**	**Mouse, male**	**Liver and lung**	**7.0**	**1.1×10^{-8}**	**Table 5-20**
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver	7.0	7.3×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Lung	7.0	4.8×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	1.0	1.6×10^{-9}	Table 5-20
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	1.0×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Tissue-specific GST-metabolism rate	Mouse, male	Lung	1.0	6.8×10^{-10}	Table 5-19
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Liver and lung	7.0	1.4×10^{-8}	Table 5-20
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	4.6×10^{-9}	Table 5-19
GST-T1$^{+/+}$	Whole-body GST metabolism rate	Mouse, male	Lung	7.0	1.0×10^{-8}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	7.0	5.5×10^{-9}	Table 5-20
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver	7.0	3.4×10^{-9}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Lung	7.0	2.6×10^{-9}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver and lung	1.0	7.9×10^{-10}	Table 5-20
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Liver	1.0	4.8×10^{-10}	Table 5-19
Mixed	Tissue-specific GST-metabolism rate	Mouse, male	Lung	1.0	3.7×10^{-10}	Table 5-19
Mixed	Whole-body GST metabolism rate	Mouse, male	Liver and lung	7.0	7.9×10^{-9}	Table 5-20
Mixed	Whole-body GST metabolism rate	Mouse, male	Liver	7.0	2.7×10^{-9}	Table 5-19
Mixed	Whole-body GST metabolism rate	Mouse, male	Lung	7.0	6.0×10^{-9}	Table 5-19
	Administered concentration (HEC)	Mouse, male	Liver		3.7×10^{-7}	Table 5-21
	Administered concentration (HEC)	Mouse, male	Lung		9.2×10^{-7}	Table 5-21
	1995 IRIS assessment[c]	Mouse, male	Liver and lung	12.7	4.7×10^{-7}	

[a]GST-T1$^{+/+}$ = homozygous, full enzyme activity; Mixed = genotypes based on a population reflecting the estimated frequency of genotypes in the current U.S. population: 20% GST-T1$^{-/-}$, 48% GST-T1$^{+/-}$, and 32% GST-T1$^{+/+}$ (Haber et al., 2002).
[b]Based on mean value of the derived distributions
Bolded value is the basis for the recommended IUR of 1×10^{-8} $\mu g/m^3$ per mg/kg-day.

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6.2.6. Uncertainties in Cancer Risk Values

The database of animal bioassays identifies the liver and lung as the most sensitive target organs for dichloromethane-induced tumor development, and there is high confidence that the dose-response data for liver and lung cancer in mice represent the best available data for derivation of human cancer risks. A dose-response relationship was seen with respect to liver cancer in mice exposed orally and with respect to liver and lung cancer in mice exposed by inhalation. Statistically significant increases in benign mammary gland tumors were observed in one study of F344 rats exposed by inhalation to 2,000 or 4,000 ppm (Mennear et al., 1988; NTP, 1986); evidence for a tumorigenic mammary gland response in Sprague-Dawley rats was limited to increased numbers of benign mammary tumors per animal at levels of 50–500 ppm (Nitschke et al., 1988a) or 500–3,500 ppm (Burek et al., 1984). An oral (gavage) study in female Sprague-Dawley rats reported an increased incidence of malignant mammary tumors, mainly adenocarcinomas (8, 6, and 18% in the control, 100, and 500 mg/kg dose groups, respectively), but the increase was not statistically significant. Data were not provided to allow an analysis that accounts for differing mortality rates (Maltoni et al., 1988). The toxicokinetic or mechanistic events that might lead to mammary gland tumor development in rats are unknown, although CYP2E1 (El-Rayes et al., 2003; Hellmold et al., 1998) and GST-T1 expression has been detected in human mammary tissue (Lehmann and Wagner, 2008). Rare CNS tumors were observed in one study in rats, a study spanning a relatively low range of exposures (0–500 ppm) (Nitschke et al., 1988a). These cancers were not seen in two other studies in rats, both involving higher doses (1,000–4,000 ppm) (NTP, 1986; Burek et al., 1984), or in a similar high-dose study in mice (NTP, 1986). The relative rarity of the tumors precludes the use of the low-dose exposure study (Nitschke et al., 1988a) in a quantitative dose-response assessment. The available epidemiologic studies provide some evidence of an association between dichloromethane and brain cancer. The available epidemiologic studies do not provide an adequate basis for the evaluation of the role of dichloromethane in breast cancer because there are currently no cohort studies with adequate statistical power and no case-control studies with adequate exposure methodology to examine this relationship.

There is uncertainty as to whether the reactivity of the toxic dichloromethane metabolites is sufficiently high enough to preclude systemic distribution. Therefore, alternative derivations of cancer risk values were performed under the assumption that high reactivity leads to complete clearance from the tissue in which the active metabolite is formed (scaling factor = 1.0). The difference in scaling factor (7.0 for allometric scaling versus 1.0) results in a 7-fold decrease in estimated cancer toxicity values. Using a whole-body GST metabolism dose metric, the resulting OSF and IUR for liver and lung cancer were approximately five-fold higher than when tissue-specific dose metrics were used (Table 5-16 and Table 5-22). The mechanistic data support the notion that reactive metabolites produced in the target tissues are not well distributed and produce deleterious effects in the metabolizing tissues soon after generation. Thus, there is

less uncertainty in the cancer risk values derived by using a tissue-specific GST metabolism dose metric compared with those derived using a whole-body GST metabolism dose metric.

Uncertainty in the ability of the PBTK models to estimate animal and human internal doses from lifetime bioassay low-level environmental exposures may affect the confidence in the cancer risk extrapolated from animal data. Uncertainties in the mouse and human model parameter values were integrated quantitatively into parameter estimation by utilizing hierarchical Bayesian methods to calibrate the models at the population level (David et al., 2006; Marino et al., 2006). The use of Monte Carlo sampling to define human model parameter distributions allowed for derivation of human distributions of dosimetry and cancer risk, providing for bounds on the recommended risk values. However, the PBTK animal models were utilized deterministically, and a sensitivity analysis was performed to identify model parameters most influential on the predictions of dose metrics used to estimate oral and inhalation cancer risks. For inhalation exposures in mice, the blood:air partition coefficient, followed closely by the first-order GST-mediated metabolism rate, had the greatest impact on the dose metric for liver cancer (mg dichloromethane metabolized via GST pathway per liter liver per day). For drinking water exposures in mice, the first-order GST-mediated metabolism rate, followed by the CYP-mediated maximum reaction velocity ($V_{max}c$) affected the liver cancer dose metric to the greatest extent. For mice inhaling dichloromethane, the lung cancer dose metric (mg dichloromethane metabolized via GST pathways per liter lung per day), like the liver cancer metric, was highly affected by the first-order GST-mediated metabolism rate and the blood:air partition coefficient. However, the lung cancer dose metric was most sensitive to the proportional yield of liver GST-mediated metabolic activity attributed to the lung. The blood:air partition coefficient was experimentally determined, lending high confidence to its value. In contrast, values for the three metabolic parameters were determined by computational optimization against data sets not directly measuring dichloromethane or its metabolites in the target/metabolizing tissues. It is uncertain how alternative values for these three parameters would affect the estimation of animal $BMDL_{10}$ values and, ultimately, IURs and OSFs. In addition, specific uncertainty remains concerning the human PBTK parameter distributions. In addition, while the structure and equations used in the existing model have been described extensively in peer-reviewed publications, uncertainty remains concerning the model structure, and specifically the potential of an alternative (dual-binding-site) CYP metabolic rate equation for dichloromethane Integration of the alternate rate equation into the PBTK modeling, and then quantitative risk assessment, will likely require several years of further research, and hence is beyond the scope of the current assessment.

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