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Toxicological Review of Hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX)

(CASRN 121-82-4)

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

March 2016

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National Center for Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Washington, DC

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ABBREVIATIONS

AAP	Army ammunition plant
ACGIH	American Conference of Governmental Industrial Hygienists
AChE	acetylcholinesterase
ADAF	age-dependent adjustment factor
AIC	Akaike's information criterion
ALP	alkaline phosphatase
ALT	alanine aminotransferase
AST	aspartate aminotransferase
atm	atmosphere
ATSDR	Agency for Toxic Substances and Disease Registry
AUC	area under the curve
BDNF	brain-derived neurotrophic factor
BHC	beta-hexachlorocyclohexane
BMC	benchmark concentration
BMCL	benchmark concentration lower confidence limit
BMD	benchmark dose
BMDL	benchmark dose lower confidence limit
BMDS	Benchmark Dose Software
BMDU	benchmark dose upper bound
BMR	benchmark response
BUN	blood urea nitrogen
BW	body weight
CASRN	Chemical Abstracts Service Registry Number
CCL	Contaminant Candidate List
CI	confidence interval
CICAD	Concise International Chemical Assessment Document
CNS	central nervous system
CSF	cerebrospinal fluid
CYP450	cytochrome P450
DAF	dosimetric adjustment factor
DDT	dichlorodiphenyltrichloroethane
d.f.	degrees of freedom
DMSO	dimethylsulfoxide
DNA	deoxyribonucleic acid
DNX	1-nitro-3,5-dinitroso-1,3,5-triazacyclohexane
DTIC	Defense Technical Information Center
EEG	electroencephalogram
EHC	Environmental Health Criteria
EPA	Environmental Protection Agency
ER	extra risk
FDA	Food and Drug Administration
FOB	functional observational battery
FUDS	Formerly Used Defense Sites
GABA	gamma-amino butyric acid
GD	gestational day
GI	gastrointestinal
GLP	good laboratory practices
HED	human equivalent dose
HERO	Health and Environmental Research Online
HGPRT	hypoxanthine-guanine phosphoribosyltransferase
HMX	octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine
IARC	International Agency for Research on Cancer
i.p.	intraperitoneal
IPCS	International Programme on Chemical Safety
IRIS	Integrated Risk Information System
IUR	inhalation unit risk
i.v.	intravenous
LDH	lactate dehydrogenase
LOAEL	lowest-observed-adverse-effect level
LOD	limit of detection
miRNA	microRNA
MNX	hexahydro-1-nitroso-3,5-dinitro-1,3,5-triazine
MOA	mode of action
MRL	Minimal Risk Level
NAPDH	nicotinamide adenine dinucleotide phosphate
NAS	National Academy of Science
NCE	normochromatic erythrocyte
NCEA	National Center for Environmental Assessment
NCI	National Cancer Institute
NCTR	National Center for Toxicological Research
NHANES	National Health and Nutrition Examination Survey
NICNAS	National Industrial Chemicals Notification and Assessment Scheme
NIEHS	National Institute of Environmental Health Sciences
NIOSH	National Institute for Occupational Safety and Health
NOAEL	no-observed-adverse-effect level
NOEL	no-observed-effect level
NPL	National Priorities List
NRC	Nuclear Regulatory Commission
NSCEP	National Service Center for Environmental Publications
NTP	National Toxicology Program
NZW	New Zealand White
OR	odds ratio

ORD	Office of Research and Development
OSF	oral slope factor
OSHA	Occupational Safety and Health Administration
PBPK	physiologically based pharmacokinetic
PCB	polychlorinated biphenyl
PCE	polychromatic erythrocyte
PEL	Permissible Exposure Limit
PND	postnatal day
POD	point of departure
PWG	Pathology Working Group
RBC	red blood cell
RDX	Royal Demolition eXplosive (hexahydro-1,3,5-trinitro-1,3,5-triazine)
REL	Recommended Exposure Limit
RfC	inhalation reference concentration
RfD	oral reference dose
SDMS	spontaneous death or moribund sacrifice
SDWA	Safe Drinking Water Act
SGOT	glutamic oxaloacetic transaminase, also known as AST
SGPT	glutamic pyruvic transaminase, also known as ALT
SLE	systemic lupus erythematosus
SS	scheduled sacrifice
TLV	Threshold Limit Value
TNT	trinitrotoluene
TNX	hexahydro-1,3,5-trinitroso-1,3,5-triazine
TSCATS	Toxic Substances Control Act Test Submissions
TWA	time-weighted average
U.S.	United States of America
UCM	Unregulated Contaminant Monitoring
UF	uncertainty factor
UF_A	animal-to-human uncertainty factor
UF_D	database deficiencies uncertainty factor
UF_H	human variation uncertainty factor
UF_L	LOAEL-to-NOAEL uncertain factor
UF_S	subchronic-to-chronic uncertainty factor
WBC	white blood cell
WHO	World Health Organization

AUTHORS | CONTRIBUTORS | REVIEWERS

Assessment Team

Louis D'Amico, Ph.D. (Assessment Manager) Todd Blessinger, Ph.D. Ravi Subramaniam, Ph.D.	U.S. EPA/ORD/NCEA Washington, DC
Christopher Brinkerhoff, Ph.D.	ORISE Postdoctoral Fellow at U.S. EPA/ORD/NCEA Currently at U.S. EPA/Office of Chemical Safety and Pollution Prevention Washington, DC

Contributors

Rob DeWoskin, Ph.D. Karen Hogan, MS Jordan Trecki, Ph.D.	U.S. EPA/ORD/NCEA Washington, DC
Belinda Hawkins, Ph.D. Scott Wesselkamper, Ph.D.	U.S. EPA/ORD/NCEA Cincinnati, OH
Tammy Stoker, Ph.D. Charles Wood, DVM, PhD, DACVP	U.S. EPA/ORD/National Health and Environmental Research Laboratory (NHEERL) Research Triangle Park, NC
Anne Loccisano, Ph.D.	ORISE Postdoctoral Fellow at U.S. EPA/ORD/NCEA Washington, DC

Production Team

Maureen Johnson Vicki Soto	U.S. EPA/ORD/NCEA Washington, DC

Contractor Support

Heather Carlson-Lynch, S.M., DABT Julie Melia, Ph.D., DABT Megan Riccardi, M.S.	SRC, Inc., North Syracuse, NY
Pam Ross, M.S. Robin Blain, Ph.D.	ICF International, Fairfax, VA

Executive Direction

Kenneth Olden, Ph.D., Sc.D., L.H.D. (Center Director) John Vandenberg, Ph.D, (National Program Director, HHRA) Lynn Flowers, Ph.D., DABT (Associate Director for Health) Vincent Cogliano, Ph.D. (IRIS Program Director) Samantha Jones, Ph.D. (IRIS Associate Director for Science) Susan Rieth, MPH (Quantitative Modeling Branch Chief)	U.S. EPA/ORD/NCEA Washington, DC

Internal Review Team

General Toxicity/Immunotoxicity/Cancer Workgroup	U.S. EPA/ORD/NCEA
Epidemiology Workgroup	
Neurotoxicity Workgroup	
Pharmacokinetics Workgroup	
Reproductive and Developmental Toxicity Workgroup	
Scoping and Problem Formulation Workgroup	
Statistics Workgroup	
Toxicity Pathways Workgroup	

Reviewers

This assessment was provided for review to scientists in EPA's program and regional offices. Comments were submitted by:

Office of the Administrator/Office of Children's Health Protection
Office of Chemical Safety and Pollution Prevention/Office of Pesticide Programs
Office of Land and Emergency Management
Office of Land and Emergency Management/Federal Facilities Forum
Office of Water
Region 2, New York, NY
Region 8, Denver, CO

This assessment was provided for review to other federal agencies and the Executive Office of the President. Comments were submitted by:

Department of Defense
Department of Energy
Department of Health and Human Services/Agency for Toxic Substances and Disease Registry
Department of Health and Human Services/National Institute of Environmental Health Sciences/National Toxicology Program
Department of Health and Human Services/National Institute for Occupational Safety and Health
National Aeronautics and Space Administration
Executive Office of the President/Council on Environmental Quality
Executive Office of the President/Office of Management and Budget

PREFACE

This Toxicological Review critically reviews the publicly available studies on hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX, Royal Demolition eXplosive, or cyclonite) in order to identify its adverse health effects and to characterize exposure-response relationships. It was prepared under the auspices of the U.S. Environmental Protection Agency (EPA) Integrated Risk Information System (IRIS) Program. This assessment updates a previous IRIS assessment of RDX that included an oral reference dose (RfD) for effects other than cancer (posted in 1988), a determination on the carcinogenicity of RDX, and a derivation of an oral slope factor (OSF) to quantify the cancer risk associated with RDX exposure (posted in 1990). New information has become available, and this assessment reviews information on all health effects by all exposure routes.

A public meeting was held in December 2013 to obtain input on preliminary materials for RDX, including draft literature searches and associated search strategies, evidence tables, and exposure-response arrays prior to the development of the IRIS assessment. All public comments provided were taken into consideration in developing the draft assessment. The complete set of public comments are available on the docket at http://www.regulations.gov (Docket ID No. EPA-HQ-ORD-2013-0430).

Organ/system-specific reference values are calculated based on nervous system, kidney/urogenital system, and male reproductive toxicity data. These reference values may be useful for cumulative risk assessments that consider the combined effect of multiple agents acting on the same biological system.

This assessment was conducted in accordance with EPA guidance, which is summarized in the Preamble to IRIS Toxicological Reviews and cited at appropriate places in this assessment. The findings of this assessment and related documents produced during its development are available on the IRIS website (http://www.epa.gov/iris). Appendices containing information on assessments by other health agencies, details of the literature search strategy, toxicokinetic information, summaries of supplementary toxicity information, and dose-response modeling are provided as Supplemental Information to this assessment (see Appendices A to D).

For additional information about this assessment or for general questions regarding IRIS, please contact EPA's IRIS Hotline at 202-566-1676 (phone), or hotline.iris@epa.gov.

Uses and Environmental Occurrence

RDX is a military munitions explosive with limited civilian uses (Gadagbui et al., 2012). In the United States, RDX is produced at Army ammunition plants (AAPs) and is not produced commercially. RDX production peaked in the 1960s; 180 million pounds per year were produced from 1969 to 1971. Yearly total production dropped to 16 million pounds in 1984 (ATSDR, 2012).

According to the U.S. EPA Chemical Data Access Tool (http://java.epa.gov/oppt_chemical_search/), the aggregate national production volume in 2012 was approximately 6.3 million pounds per year.

RDX can be released into environmental media (air, water, soil) as a result of waste generated during manufacture, packing, or disposal of the pure product, or use and disposal of RDX-containing munitions (ATSDR, 2012; Gadagbui et al., 2012; ATSDR, 1999, 1993, 1992). RDX is mobile in soil; leaching into groundwater has been reported in samples from military facilities (Best et al., 1999a; Godejohann et al., 1998; Bart et al., 1997; Steuckart et al., 1994; Spanggord et al., 1980a). RDX transport in soil is generally through dissolution by precipitation and subsequent downward movement, including migration to groundwater aquifers, and not much via surface runoff (U.S. EPA, 2012c). An extensive discussion of RDX properties and fate and transport is available in U.S. EPA (2012c). Detectable levels of RDX have been observed in plants irrigated or grown with RDX-contaminated water (Best et al., 1999b; Simini and Checkai, 1996; Harvey et al., 1991). RDX has also been detected in indoor air samples from military facilities where RDX is produced (Bishop et al., 1988).

Exposures to RDX among the general population are likely to be confined to individuals in or around active or formerly-used military facilities where RDX is or was produced, stored, or used. Oral, inhalation, and dermal routes of exposure may be relevant.

As of 2015, RDX was detected in surface water, groundwater, sediment, or soil at 34 current U.S. EPA National Priorities List (NPL) sites. The NPL serves as a list of sites with known or threatened releases of hazardous substances, pollutants, or contaminants throughout the United States and its territories. The NPL aids the Agency in identifying the most serious sites that may warrant cleanup. The majority of the NPL sites where RDX was listed are associated with military facilities. Based on Department of Defense records, Gadagbui et al. (2012) reported that RDX contamination is present on 76 active military sites, 9 closed sites, and 15 sites under the Formerly Used Defense Sites (FUDS) program. Not all sites under the FUDS program have been sampled, and additional sites with RDX contamination in this program could be identified.

As of 2015, RDX was not regulated under the Safe Drinking Water Act (SDWA), although it was included as a contaminant to be monitored under the Unregulated Contaminant Monitoring (UCM) Rule by EPA's Office of Water from 2007 to 2011. Contaminants included in the UCM program are suspected of being present in drinking water, but do not have existing health-based standards set under the SDWA. RDX has also been included on the Office of Water's Drinking Water Contaminant Candidate List (CCL) since the initial listing was published in 1998. The presence of a chemical on the list suggests that it is known or anticipated to occur in public water systems.

Assessments by Other National and International Health Agencies

Toxicity information on RDX has been evaluated by the Agency for Toxic Substances and Disease Registry (ATSDR), National Institute for Occupational Safety and Health (NIOSH), Occupational Safety and Health Administration (OSHA), and Australian National Industrial Chemicals Notification and Assessment Scheme (NICNAS). The results of these assessments (as of

2015) are presented in Appendix A of the Supplemental Information. It is important to recognize that the assessments performed by other health agencies may have been prepared for different purposes and may utilize different methods. In addition, newer studies may be included in the IRIS assessment.

PREAMBLE TO IRIS TOXICOLOGICAL REVIEWS

Note: The Preamble to IRIS assessments is being revised based on comments received from external peer reviewers and the public, and based on IRIS Program experience with the implementation of systematic review methods. Subsequent drafts of the RDX assessment will include the revised Preamble.

1. Scope of the IRIS Program

Soon after the EPA was established in 1970, it was at the forefront of developing risk assessment as a science and applying it in decisions to protect human health and the environment. The Clean Air Act, for example, mandates that the EPA provide "an ample margin of safety to protect public health"; the Safe Drinking Water Act, that "no adverse effects on the health of persons may reasonably be anticipated to occur, allowing an adequate margin of safety." Accordingly, the EPA uses information on the adverse effects of chemicals and on exposure levels below which these effects are not anticipated to occur.

IRIS assessments critically review the publicly available studies to identify adverse health effects from exposure to chemicals and to characterize exposure-response relationships. In terms set forth by the National Research Council (NRC, 1983), IRIS assessments cover the hazard identification and dose-response assessment steps of risk assessment, not the exposure assessment or risk characterization steps that are conducted by the EPA's program and regional offices and by other federal, state, and local health agencies that evaluate risk in specific populations and exposure scenarios. IRIS assessments are distinct from and do not address political, economic, and technical considerations that influence the design and selection of risk management alternatives.

An IRIS assessment may cover a single chemical, a group of structurally or toxicologically related chemicals, or a complex mixture. These agents may be found in air, water, soil, or sediment. Exceptions are chemicals currently used exclusively as pesticides, ionizing and non-ionizing radiation, and criteria air pollutants listed under Section 108 of the Clean Air Act (carbon monoxide, lead, nitrogen oxides, ozone, particulate matter, and sulfur oxides).

Periodically, the IRIS Program asks other EPA programs and regions, other federal agencies, state health agencies, and the general public to nominate chemicals and mixtures for future assessment or reassessment. Agents may be considered for reassessment as significant new studies are published. Selection is based on program and regional office priorities and on availability of adequate information to evaluate the potential for adverse effects. Other agents may also be assessed in response to an urgent public health need.

2. Process for developing and peer-reviewing IRIS assessments

The process for developing IRIS assessments (revised in May 2009 and enhanced in July 2013) involves critical analysis of the pertinent studies, opportunities for public input, and multiple levels of scientific review. The EPA revises draft assessments after each review, and external drafts and comments become part of the public record (U.S. EPA, 2009).

Before beginning an assessment, the IRIS Program discusses the scope with other EPA programs and regions to ensure that the

assessment will meet their needs. Then a public meeting on problem formulation invites discussion of the key issues and the studies and analytical approaches that might contribute to their resolution.

Step 1. Development of a draft Toxicological Review. The draft assessment considers all pertinent publicly available studies and applies consistent criteria to evaluate study quality, identify health effects, identify mechanistic events and pathways, integrate the evidence of causation for each effect, and derive toxicity values. A public meeting prior to the integration of evidence and derivation of toxicity values promotes public discussion of the literature search, evidence, and key issues.

Step 2. Internal review by scientists in EPA programs and regions. The draft assessment is revised to address the comments from within the EPA.

Step 3. Interagency science consultation with other federal agencies and the Executive Offices of the President. The draft assessment is revised to address the interagency comments. The science consultation draft, interagency comments, and the EPA's response to major comments become part of the public record.

Step 4. Public review and comment, followed by external peer review. The EPA releases the draft assessment for public review and comment. A public meeting provides an opportunity to discuss the assessment prior to peer review. Then the EPA releases a draft for external peer review. The peer review meeting is open to the public and includes time for oral public comments. The peer reviewers assess whether the evidence has been assembled and evaluated according to guidelines and whether the conclusions are justified by the evidence. The peer review draft, written public comments, and peer review report become part of the public record.

Step 5. Revision of draft Toxicological Review and development of draft IRIS summary. The draft assessment is revised to reflect the peer review comments, public comments, and newly published studies that are critical to the conclusions of the assessment. The disposition of peer review comments and public comments becomes part of the public record.

Step 6. Final EPA review and interagency science discussion with other federal agencies and the Executive Offices of the President The draft assessment and summary are revised to address the EPA and interagency comments. The science discussion draft, written interagency comments, and EPA's response to major comments become part of the public record.

Step 7. Completion and posting. The Toxicological Review and IRIS summary are posted on the IRIS website (http://www.epa.gov/iris/).

The remainder of this Preamble addresses step 1, the development of a draft Toxicological Review. IRIS assessments follow standard practices of evidence evaluation and peer review, many of which are discussed in EPA guidelines (U.S. EPA, 2005a, b, 2000b, 1998, 1996, 1991, 1986a, b) and other methods (U.S. EPA, 2012a, b, 2011, 2006a, b, 2002, 1994). Transparent application of scientific judgment is of paramount importance. To provide a harmonized approach across IRIS assessments, this Preamble summarizes concepts from these guidelines and emphasizes principles of general applicability.

3. Identifying and selecting pertinent studies

3.1. Identifying studies

Before beginning an assessment, the EPA conducts a comprehensive search of the primary scientific literature. The literature search follows standard practices and includes the PubMed and ToxNet databases of the National Library of Medicine, Web of Science, and other databases listed in the EPA's HERO system (Health and Environmental Research Online, http://hero.epa.gov/). Searches for information on mechanisms of toxicity are inherently specialized and may include studies on other agents that act through related mechanisms.

Each assessment specifies the search strategies, keywords, and cut-off dates of its literature searches. The EPA posts the results of the literature search on the IRIS website and requests information from the public on additional studies and ongoing research.

The EPA also considers studies received through the IRIS Submission Desk and studies (typically unpublished) submitted under the Toxic Substances Control Act or the Federal Insecticide, Fungicide, and Rodenticide Act. Material submitted as Confidential Business Information is considered only if it includes health and safety data that can be publicly released. If a study that may be critical to the conclusions of the assessment has not been peer-reviewed, the EPA will have it peer-reviewed.

The EPA also examines the toxicokinetics of the agent to identify other chemicals (for example, major metabolites of the agent) to include in the assessment if adequate information is available, in order to more fully explain the toxicity of the agent and to suggest dose metrics for subsequent modeling.

In assessments of chemical mixtures, mixture studies are preferred for their ability to reflect interactions among components.

The literature search seeks, in decreasing order of preference (U.S. EPA, 2000b, §2.2; 1986b, §2.1)]:

- Studies of the mixture being assessed.
- Studies of a sufficiently similar mixture. In evaluating similarity, the assessment considers the alteration of mixtures in the environment through partitioning and transformation.
- Studies of individual chemical components of the mixture, if there are not adequate studies of sufficiently similar mixtures.

3.2. Selecting pertinent epidemiologic studies

Study design is the key consideration for selecting pertinent epidemiologic studies from the results of the literature search.

- Cohort studies, case-control studies, and some population-based surveys (for example, NHANES) provide the strongest epidemiologic evidence, especially if they collect information about individual exposures and effects.
- Ecological studies (geographic correlation studies) relate exposures and effects by geographic area. They can provide strong evidence if there are large exposure contrasts between geographic areas, relatively little exposure variation within study areas, and population migration is limited.
- Case reports of high or accidental exposure lack definition of the population at risk and the expected number of cases. They can provide information about a rare effect or about the relevance of analogous results in animals.

The assessment briefly reviews ecological studies and case reports but reports details only if they suggest effects not identified by other studies.

3.3. Selecting pertinent experimental studies

Exposure route is a key design consideration for selecting pertinent

experimental animal studies or human clinical studies.

- Studies of oral, inhalation, or dermal exposure involve passage through an absorption barrier and are considered most pertinent to human environmental exposure.
- Injection or implantation studies are often considered less pertinent but may provide valuable toxicokinetic or mechanistic information. They also may be useful for identifying effects in animals if deposition or absorption is problematic (for example, for particles and fibers).

Exposure duration is also a key design consideration for selecting pertinent experimental animal studies.

- Studies of effects from chronic exposure are most pertinent to lifetime human exposure.
- Studies of effects from less-than-chronic exposure are pertinent but less preferred for identifying effects from lifetime human exposure. Such studies may be indicative of effects from less-than-lifetime human exposure.

Short-duration studies involving animals or humans may provide toxicokinetic or mechanistic information.

For developmental toxicity and reproductive toxicity, irreversible effects may result from a brief exposure during a critical period of development. Accordingly, specialized study designs are used for these effects (U.S. EPA, 2006b, 1998, 1996, 1991).

4. Evaluating the quality of individual studies

After the subsets of pertinent epidemiologic and experimental studies have been selected from the literature searches, the assessment evaluates the quality of each individual study. This evaluation considers the design, methods, conduct, and documentation of each study, but not whether the results are positive, negative, or null. The objective is to identify the stronger, more informative studies based on a uniform evaluation of quality characteristics across studies of similar design.

4.1. Evaluating the quality of epidemiologic studies

The assessment evaluates design and methodological aspects that can increase or decrease the weight given to each epidemiologic study in the overall evaluation (U.S. EPA, 2005a, 1998, 1996, 1994, 1991):

- Documentation of study design, methods, population characteristics, and results.
- Definition and selection of the study group and comparison group.
- Ascertainment of exposure to the chemical or mixture.
- Ascertainment of disease or health effect.
- Duration of exposure and follow-up and adequacy for assessing the occurrence of effects.
- Characterization of exposure during critical periods.
- Sample size and statistical power to detect anticipated effects.
- Participation rates and potential for selection bias as a result of the achieved participation rates.
- Measurement error (can lead to misclassification of exposure, health outcomes, and other factors) and other types of information bias.
- Potential confounding and other sources of bias addressed in the study design or in the analysis of results. The basis for consideration of confounding is a reasonable expectation that the confounder is related to both exposure and

outcome and is sufficiently prevalent to result in bias.

For developmental toxicity, reproductive toxicity, neurotoxicity, and cancer there is further guidance on the nuances of evaluating epidemiologic studies of these effects (U.S. EPA, 2005a, 1998, 1996, 1991).

4.2. Evaluating the quality of experimental studies

The assessment evaluates design and methodological aspects that can increase or decrease the weight given to each experimental animal study, in-vitro study, or human clinical study (U.S. EPA, 2005a, 1998, 1996, 1991). Research involving human subjects is considered only if conducted according to ethical principles.

- Documentation of study design, animals or study population, methods, basic data, and results.
- Nature of the assay and validity for its intended purpose.
- Characterization of the nature and extent of impurities and contaminants of the administered chemical or mixture.
- Characterization of dose and dosing regimen (including age at exposure) and their adequacy to elicit adverse effects, including latent effects.
- Sample sizes and statistical power to detect dose-related differences or trends.
- Ascertainment of survival, vital signs, disease or effects, and cause of death.
- Control of other variables that could influence the occurrence of effects.

The assessment uses statistical tests to evaluate whether the observations may be due to chance. The standard for determining statistical significance of a response is a trend test or comparison of outcomes in the exposed groups against those of concurrent controls. In some situations, examination of historical control data from the same laboratory within a few years of the study may improve the analysis. For an uncommon effect that is not statistically significant compared with concurrent controls, historical controls may show that the effect is unlikely to be due to chance. For a response that appears significant against a concurrent control response that is unusual, historical controls may offer a different interpretation (U.S. EPA, 2005a, §2.2.2.1.3).

For developmental toxicity, reproductive toxicity, neurotoxicity, and cancer there is further guidance on the nuances of evaluating experimental studies of these effects (U.S. EPA, 2005a, 1998, 1996, 1991). In multi-generation studies, agents that produce developmental effects at doses that are not toxic to the maternal animal are of special concern. Effects that occur at doses associated with mild maternal toxicity are not assumed to result only from maternal toxicity. Moreover, maternal effects may be reversible, while effects on the offspring may be permanent (U.S. EPA, 1998, §3.1.2.4.5.4; 1991, §3.1.1.4),.

4.3. Reporting study results

The assessment uses evidence tables to present the design and key results of pertinent studies. There may be separate tables for each site of toxicity or type of study.

If a large number of studies observe the same effect, the assessment considers the study quality characteristics in this section to identify the strongest studies or types of study. The tables present details from these studies, and the assessment explains the reasons for not reporting details of other studies or groups of studies that do not add new information. Supplemental information provides references to all studies considered, including those not summarized in the tables.

The assessment discusses strengths and limitations that affect the interpretation of each study. If the interpretation of a study in the assessment differs from that of the study authors, the assessment discusses the basis for the difference.

As a check on the selection and evaluation of pertinent studies, the EPA asks peer reviewers to identify studies that were not adequately considered.

5. Evaluating the overall evidence of each effect

5.1. Concepts of causal inference

For each health effect, the assessment evaluates the evidence as a whole to determine whether it is reasonable to infer a causal association between exposure to the agent and the occurrence of the effect. This inference is based on information from pertinent human studies, animal studies, and mechanistic studies of adequate quality. Positive, negative, and null results are given weight according to study quality.

Causal inference involves scientific judgment, and the considerations are nuanced and complex. Several health agencies have developed frameworks for causal inference, among them the U.S. Surgeon General (CDC, 2004; HEW, 1964), the International Agency for Research on Cancer (IARC, 2006), the Institute of Medicine (IOM, 2008), and the (U.S. EPA (2010), §1.6; 2005a), §2.5). Although developed for different purposes, the frameworks are similar in nature and provide an established structure and language for causal inference. Each considers aspects of an association that suggest causation, discussed by Hill (1965) and elaborated by Rothman and Greenland (1998), and (U.S. EPA (2005a), §2.2.1.7; 1994), Appendix C).

Strength of association: The finding of a large relative risk with narrow confidence intervals strongly suggests that an association is not due to chance, bias, or other factors. Modest relative risks, however, may reflect a small range of exposures, an agent of low potency, an increase in an effect that is common, exposure misclassification, or other sources of bias.

Consistency of association: An inference of causation is strengthened if elevated risks are observed in independent studies of different populations and exposure scenarios. Reproducibility of findings constitutes one of the strongest arguments for causation. Discordant results sometimes reflect differences in study design, exposure, or confounding factors.

Specificity of association: As originally intended, this refers to one cause associated with one effect. Current understanding that many agents cause multiple effects and many effects have multiple causes make this a less informative aspect of causation, unless the effect is rare or unlikely to have multiple causes.

Temporal relationship: A causal interpretation requires that exposure precede development of the effect.

Biologic gradient (exposure-response relationship): Exposure-response relationships strongly suggest causation. A monotonic increase is not the only pattern consistent with causation. The presence of an exposure-response gradient also weighs against bias and confounding as the source of an association.

Biologic plausibility: An inference of causation is strengthened by data demonstrating plausible biologic mechanisms, if available. Plausibility may reflect subjective prior beliefs if there is insufficient understanding of the biologic process involved.

Coherence: An inference of causation is strengthened by supportive results from animal experiments, toxicokinetic studies, and short-term tests. Coherence may also be found in other lines of evidence, such as changing disease patterns in the population.

"Natural experiments": A change in exposure that brings about a change in

disease frequency provides strong evidence, as it tests the hypothesis of causation. An example would be an intervention to reduce exposure in the workplace or environment that is followed by a reduction of an adverse effect.

Analogy: Information on structural analogues or on chemicals that induce similar mechanistic events can provide insight into causation.

These considerations are consistent with guidelines for systematic reviews that evaluate the quality and weight of evidence. Confidence is increased if the magnitude of effect is large, if there is evidence of an exposure-response relationship, or if an association was observed and the plausible biases would tend to decrease the magnitude of the reported effect. Confidence is decreased for study limitations, inconsistency of results, indirectness of evidence, imprecision, or reporting bias (Guyatt et al., 2008b; Guyatt et al., 2008a).

5.2. Evaluating evidence in humans

For each effect, the assessment evaluates the evidence from the epidemiologic studies as a whole. The objective is to determine whether a credible association has been observed and, if so, whether that association is consistent with causation. In doing this, the assessment explores alternative explanations (such as chance, bias, and confounding) and draws a conclusion about whether these alternatives can satisfactorily explain any observed association.

To make clear how much the epidemiologic evidence contributes to the overall weight of the evidence, the assessment may select a standard descriptor to characterize the epidemiologic evidence of association between exposure to the agent and occurrence of a health effect.

Sufficient epidemiologic evidence of an association consistent with causation: The evidence establishes a causal association for which alternative explanations such as chance, bias, and confounding can be ruled out with reasonable confidence.

Suggestive epidemiologic evidence of an association consistent with causation: The evidence suggests a causal association but chance, bias, or confounding cannot be ruled out as explaining the association.

Inadequate epidemiologic evidence to infer a causal association: The available studies do not permit a conclusion regarding the presence or absence of an association.

Epidemiologic evidence consistent with no causal association: Several adequate studies covering the full range of human exposures and considering susceptible populations, and for which alternative explanations such as bias and confounding can be ruled out, are mutually consistent in not finding an association.

5.3. Evaluating evidence in animals

For each effect, the assessment evaluates the evidence from the animal experiments as a whole to determine the extent to which they indicate a potential for effects in humans. Consistent results across various species and strains increase confidence that similar results would occur in humans. Several concepts discussed by Hill (1965) are pertinent to the weight of experimental results: consistency of response, dose-response relationships, strength of response, biologic plausibility, and coherence (U.S. EPA, 2005a, §2.2.1.7; 1994, Appendix C).

In weighing evidence from multiple experiments, U.S. EPA (2005a), §2.5 distinguishes:

Conflicting evidence (that is, mixed positive and negative results in the same sex and strain using a similar study protocol) from

Differing results (that is, positive results and negative results are in different sexes or strains or use different study protocols).

Negative or null results do not invalidate positive results in a different experimental system. The EPA regards all as valid observations and looks to explain differing results using mechanistic information (for example, physiologic or metabolic differences across test systems) or methodological differences (for example, relative sensitivity of the tests, differences in dose levels, insufficient sample size, or timing of dosing or data collection).

It is well established that there are critical periods for some developmental and reproductive effects (U.S. EPA, 2006b, 2005a, b, 1998, 1996, 1991). Accordingly, the assessment determines whether critical periods have been adequately investigated. Similarly, the assessment determines whether the database is adequate to evaluate other critical sites and effects.

In evaluating evidence of genetic toxicity:

- Demonstration of gene mutations, chromosome aberrations, or aneuploidy in humans or experimental mammals (*in vivo*) provides the strongest evidence.
- This is followed by positive results in lower organisms or in cultured cells (*in vitro*) or for other genetic events.
- Negative results carry less weight, partly because they cannot exclude the possibility of effects in other tissues (IARC, 2006).

For germ-cell mutagenicity, the EPA has defined categories of evidence, ranging from positive results of human germ-cell mutagenicity to negative results for all effects of concern (U.S. EPA, 1986a, §2.3).

5.4. Evaluating mechanistic data

Mechanistic data can be useful in answering several questions.

- The biologic plausibility of a causal interpretation of human studies.
- The generalizability of animal studies to humans.
- The susceptibility of particular populations or lifestages.

The focus of the analysis is to describe, if possible, mechanistic pathways that lead to a health effect. These pathways encompass:

- *Toxicokinetic processes* of absorption, distribution, metabolism, and elimination that lead to the formation of an active agent and its presence at the site of initial biologic interaction.
- *Toxicodynamic processes* that lead to a health effect at this or another site (also known as a *mode of action*).

For each effect, the assessment discusses the available information on its *modes of action* and associated *key events* (*key events* being empirically observable, necessary precursor steps or biologic markers of such steps; *mode of action* being a series of key events involving interaction with cells, operational and anatomic changes, and resulting in disease). Pertinent information may also come from studies of metabolites or of compounds that are structurally similar or that act through similar mechanisms. Information on mode of action is not required for a conclusion that the agent is causally related to an effect (U.S. EPA, 2005a, §2.5).

The assessment addresses several questions about each hypothesized mode of action (U.S. EPA, 2005a, §2.4.3.4).

1) **Is the hypothesized mode of action sufficiently supported in test animals?** Strong support for a key event being necessary to a mode of action can come from experimental challenge to the hypothesized mode of action, in which studies that suppress a key event observe suppression of the effect. Support for a mode of action is meaningfully strengthened by consistent results in different experimental models, much

more so than by replicate experiments in the same model. The assessment may consider various aspects of causation in addressing this question.

2) **Is the hypothesized mode of action relevant to humans?** The assessment reviews the key events to identify critical similarities and differences between the test animals and humans. Site concordance is not assumed between animals and humans, though it may hold for certain effects or modes of action. Information suggesting quantitative differences in doses where effects would occur in animals or humans is considered in the dose-response analysis. Current levels of human exposure are not used to rule out human relevance, as IRIS assessments may be used in evaluating new or unforeseen circumstances that may entail higher exposures.

3) **Which populations or lifestages can be particularly susceptible to the hypothesized mode of action?** The assessment reviews the key events to identify populations and lifestages that might be susceptible to their occurrence. Quantitative differences may result in separate toxicity values for susceptible populations or lifestages.

The assessment discusses the likelihood that an agent operates through multiple modes of action. An uneven level of support for different modes of action can reflect disproportionate resources spent investigating them (U.S. EPA, 2005a, §2.4.3.3). It should be noted that in clinical reviews, the credibility of a series of studies is reduced if evidence is limited to studies funded by one interested sector (Guyatt et al., 2008a).

For cancer, the assessment evaluates evidence of a mutagenic mode of action to guide extrapolation to lower doses and consideration of susceptible lifestages. Key data include the ability of the agent or a metabolite to react with or bind to DNA, positive results in multiple test systems, or similar properties and structure-activity relationships to mutagenic carcinogens (U.S. EPA, 2005a ,§2.3.5).

5.5. Characterizing the overall weight of the evidence

After evaluating the human, animal, and mechanistic evidence pertinent to an effect, the assessment answers the question: Does the agent cause the adverse effect? (NRC, 2009, 1983). In doing this, the assessment develops a narrative that integrates the evidence pertinent to causation. To provide clarity and consistency, the narrative includes a standard hazard descriptor. For example, the following standard descriptors combine epidemiologic, experimental, and mechanistic evidence of carcinogenicity (U.S. EPA, 2005a, §2.5).

Carcinogenic to humans: There is convincing epidemiologic evidence of a causal association (that is, there is reasonable confidence that the association cannot be fully explained by chance, bias, or confounding); or there is strong human evidence of cancer or its precursors, extensive animal evidence, identification of key precursor events in animals, and strong evidence that they are anticipated to occur in humans.

Likely to be carcinogenic to humans: The evidence demonstrates a potential hazard to humans but does not meet the criteria for *carcinogenic*. There may be a plausible association in humans, multiple positive results in animals, or a combination of human, animal, or other experimental evidence.

Suggestive evidence of carcinogenic potential: The evidence raises concern for effects in humans but is not sufficient for a stronger conclusion. This descriptor covers a range of evidence, from a positive result in the only available study to a single positive result in an extensive database that includes negative results in other species.

Inadequate information to assess carcinogenic potential: No other descriptors apply. *Conflicting evidence* can be classified as *inadequate information* if all positive results are opposed by negative studies of equal quality in the same sex and strain. *Differing results*, however, can be classified as *suggestive evidence* or as *likely to be carcinogenic*.

Not likely to be carcinogenic to humans: There is robust evidence for concluding that there is no basis for concern. There may be no effects in both sexes of at least two appropriate animal species; positive animal results and strong, consistent evidence that each mode of action in animals does not operate in humans; or convincing evidence that effects are not likely by a particular exposure route or below a defined dose.

Multiple descriptors may be used if there is evidence that carcinogenic effects differ by dose range or exposure route (U.S. EPA, 2005a, §2.5).

Another example of standard descriptors comes from the EPA's Integrated Science Assessments, which evaluate causation for the effects of the criteria pollutants in ambient air (U.S. EPA, 2010).

Causal relationship: Sufficient evidence to conclude that there is a causal relationship. Observational studies cannot be explained by plausible alternatives, or they are supported by other lines of evidence, for example, animal studies or mechanistic information.

Likely to be a causal relationship: Sufficient evidence that a causal relationship is likely, but important uncertainties remain. For example, observational studies show an association but co-exposures are difficult to address or other lines of evidence are limited or inconsistent; or multiple animal studies from different laboratories demonstrate effects and there are limited or no human data.

Suggestive of a causal relationship: At least one high-quality epidemiologic study shows an association but other studies are inconsistent.

Inadequate to infer a causal relationship: The studies do not permit a conclusion regarding the presence or absence of an association.

Not likely to be a causal relationship: Several adequate studies, covering the full range of human exposure and considering susceptible populations, are mutually consistent in not showing an effect at any level of exposure.

The EPA is investigating and may on a trial basis use these or other standard descriptors to characterize the overall weight of the evidence for effects other than cancer.

6. Selecting studies for derivation of toxicity values

For each effect where there is credible evidence of an association with the agent, the assessment derives toxicity values if there are suitable epidemiologic or experimental data. The decision to derive toxicity values may be linked to the hazard descriptor.

Dose-response analysis requires quantitative measures of dose and response. Then, other factors being equal:

- Epidemiologic studies are preferred over animal studies, if quantitative measures of exposure are available and effects can be attributed to the agent.
- Among experimental animal models, those that respond most like humans are preferred, if the comparability of response can be determined.
- Studies by a route of human environmental exposure are preferred, although a validated

toxicokinetic model can be used to extrapolate across exposure routes.

- Studies of longer exposure duration and follow-up are preferred, to minimize uncertainty about whether effects are representative of lifetime exposure.
- Studies with multiple exposure levels are preferred for their ability to provide information about the shape of the exposure-response curve.
- Studies with adequate power to detect effects at lower exposure levels are preferred, to minimize the extent of extrapolation to levels found in the environment.

Studies with non-monotonic exposure-response relationships are not necessarily excluded from the analysis. A diminished effect at higher exposure levels may be satisfactorily explained by factors such as competing toxicity, saturation of absorption or metabolism, exposure misclassification, or selection bias.

If a large number of studies are suitable for dose-response analysis, the assessment considers the study characteristics in this section to focus on the most informative data. The assessment explains the reasons for not analyzing other groups of studies. As a check on the selection of studies for dose-response analysis, the EPA asks peer reviewers to identify studies that were not adequately considered.

7. Deriving toxicity values

7.1. General framework for dose-response analysis

The EPA uses a two-step approach that distinguishes analysis of the observed dose-response data from inferences about lower doses (U.S. EPA, 2005a, §3).

Within the observed range, the preferred approach is to use modeling to incorporate a wide range of data into the analysis. The modeling yields a *point of departure* (an exposure level near the lower end of the observed range, without significant extrapolation to lower doses) (Sections 7.2-7.3).

Extrapolation to lower doses considers what is known about the modes of action for each effect (Sections 7.4-7.5). If response estimates at lower doses are not required, an alternative is to derive *reference values*, which are calculated by applying factors to the point of departure in order to account for sources of uncertainty and variability (Section 7.6).

For a group of agents that induce an effect through a common mode of action, the dose-response analysis may derive a *relative potency factor* for each agent. A full dose-response analysis is conducted for one well-studied *index chemical* in the group, then the potencies of other members are expressed in relative terms based on relative toxic effects, relative absorption or metabolic rates, quantitative structure-activity relationships, or receptor binding characteristics (U.S. EPA, 2005a, §3.2.6; 2000b, §4.4).

Increasingly, the EPA is basing toxicity values on combined analyses of multiple data sets or multiple responses. The EPA also considers multiple dose-response approaches if they can be supported by robust data.

7.2. Modeling dose to sites of biologic effects

The preferred approach for analysis of dose is toxicokinetic modeling because of its ability to incorporate a wide range of data. The preferred dose metric would refer to the active agent at the site of its biologic effect or to a close, reliable surrogate measure. The active agent may be the administered chemical or a metabolite. Confidence in the use of a toxicokinetic model depends on the robustness of its validation process and on the results of sensitivity analyses (U.S. EPA, 2006a; 2005a, §3.1; 1994, §4.3).

Because toxicokinetic modeling can require many parameters and more data than are typically available, the EPA has developed

standard approaches that can be applied to typical data sets. These standard approaches also facilitate comparison across exposure patterns and species.

- Intermittent study exposures are standardized to a daily average over the duration of exposure. For chronic effects, daily exposures are averaged over the lifespan. Exposures during a critical period, however, are not averaged over a longer duration (U.S. EPA, 2005a, §3.1.1; 1991, §3.2).
- Doses are standardized to equivalent human terms to facilitate comparison of results from different species.
- Oral doses are scaled allometrically using $mg/kg^{3/4}$-day as the equivalent dose metric across species. Allometric scaling pertains to equivalence across species, not across lifestages, and is not used to scale doses from adult humans or mature animals to infants or children (U.S. EPA, 2011; 2005a, §3.1.3).
- Inhalation exposures are scaled using dosimetry models that apply species-specific physiologic and anatomic factors and consider whether the effect occurs at the site of first contact or after systemic circulation (U.S. EPA, 2012a; 1994, §3).

It can be informative to convert doses across exposure routes. If this is done, the assessment describes the underlying data, algorithms, and assumptions (U.S. EPA, 2005a, §3.1.4).

In the absence of study-specific data on, for example, intake rates or body weight, the EPA has developed recommended values for use in dose-response analysis (U.S. EPA, 1988).

7.3. Modeling response in the range of observation

Toxicodynamic ("biologically based") modeling can incorporate data on biologic processes leading to an effect. Such models require sufficient data to ascertain a mode of action and to quantitatively support model parameters associated with its key events. Because different models may provide equivalent fits to the observed data but diverge substantially at lower doses, critical biologic parameters should be measured from laboratory studies, not by model fitting. Confidence in the use of a toxicodynamic model depends on the robustness of its validation process and on the results of sensitivity analyses. Peer review of the scientific basis and performance of a model is essential (U.S. EPA, 2005a, §3.2.2).

Because toxicodynamic modeling can require many parameters and more knowledge and data than are typically available, the EPA has developed a standard set of empirical ("curve-fitting") models (http://www.epa.gov/ncea/bmds/) that can be applied to typical data sets, including those that are nonlinear. The EPA has also developed guidance on modeling dose-response data, assessing model fit, selecting suitable models, and reporting modeling results (U.S. EPA, 2012b). Additional judgment or alternative analyses are used if the procedure fails to yield reliable results, for example, if the fit is poor, modeling may be restricted to the lower doses, especially if there is competing toxicity at higher doses (U.S. EPA, 2005a, §3.2.3).

Modeling is used to derive a point of departure (U.S. EPA, 2012b; 2005a, §3.2.4). (See Section 7.6 for alternatives if a point of departure cannot be derived by modeling.):

- If linear extrapolation is used, selection of a response level corresponding to the point of departure is not highly influential, so standard values near the low end of the observable range are generally used (for example, 10% extra risk for cancer bioassay data, 1% for epidemiologic data, lower for rare cancers).
- For nonlinear approaches, both statistical and biologic considerations are taken into account.

- For dichotomous data, a response level of 10% extra risk is generally used for minimally adverse effects, 5% or lower for more severe effects.
- For continuous data, a response level is ideally based on an established definition of biologic significance. In the absence of such definition, one control standard deviation from the control mean is often used for minimally adverse effects, one-half standard deviation for more severe effects.

The point of departure is the 95% lower bound on the dose associated with the selected response level.

7.4. Extrapolating to lower doses and response levels

The purpose of extrapolating to lower doses is to estimate responses at exposures below the observed data. Low-dose extrapolation, typically used for cancer data, considers what is known about modes of action (U.S. EPA, 2005a, §3.3.1 and §3.3.2).

1) If a biologically based model has been developed and validated for the agent, extrapolation may use the fitted model below the observed range if significant model uncertainty can be ruled out with reasonable confidence.

2) Linear extrapolation is used if the dose-response curve is expected to have a linear component below the point of departure. This includes:

   - Agents or their metabolites that are DNA-reactive and have direct mutagenic activity.
   - Agents or their metabolites for which human exposures or body burdens are near doses associated with key events leading to an effect.

   Linear extrapolation is also used when data are insufficient to establish mode of action and when scientifically plausible. The result of linear extrapolation is described by an oral slope factor or an inhalation unit risk, which is the slope of the dose-response curve at lower doses or concentrations, respectively.

3) Nonlinear models are used for extrapolation if there are sufficient data to ascertain the mode of action and to conclude that it is not linear at lower doses, and the agent does not demonstrate mutagenic or other activity consistent with linearity at lower doses. Nonlinear approaches generally should not be used in cases where mode of action has not ascertained. If nonlinear extrapolation is appropriate but no model is developed, an alternative is to calculate reference values.

4) Both linear and nonlinear approaches may be used if there a multiple modes of action. For example, modeling to a low response level can be useful for estimating the response at doses where a high-dose mode of action would be less important.

If linear extrapolation is used, the assessment develops a candidate slope factor or unit risk for each suitable data set. These results are arrayed, using common dose metrics, to show the distribution of relative potency across various effects and experimental systems. The assessment then derives or selects an overall slope factor and an overall unit risk for the agent, considering the various dose-response analyses, the study preferences discussed in Section 6, and the possibility of basing a more robust result on multiple data sets.

7.5. Considering susceptible populations and lifestages

The assessment analyzes the available information on populations and lifestages that may be particularly susceptible to each effect. A tiered approach is used (U.S. EPA, 2005a, §3.5).

1) If an epidemiologic or experimental study reports quantitative results for a susceptible population or lifestage, these data are analyzed to derive separate toxicity values for susceptible individuals.

2) If data on risk-related parameters allow comparison of the general population and susceptible individuals, these data are used to adjust the general-population toxicity values for application to susceptible individuals.

3) In the absence of chemical-specific data, the EPA has developed *age-dependent adjustment factors* for early-life exposure to potential carcinogens that have a mutagenic mode of action. There is evidence of early-life susceptibility to various carcinogenic agents, but most epidemiologic studies and cancer bioassays do not include early-life exposure. To address the potential for early-life susceptibility, the EPA recommends (U.S. EPA, 2005b, §5):

   - 10-fold adjustment for exposures before age 2 years.
   - 3-fold adjustment for exposures between ages 2 and 16 years.

7.6. Reference values and uncertainty factors

An *oral reference dose* or an *inhalation reference concentration* is an estimate of an exposure (including in susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime (U.S. EPA, 2002, §4.2). Reference values are typically calculated for effects other than cancer and for suspected carcinogens if a well characterized mode of action indicates that a necessary key event does not occur below a specific dose. Reference values provide no information about risks at higher exposure levels.

The assessment characterizes effects that form the basis for reference values as adverse, considered to be adverse, or a precursor to an adverse effect. For developmental toxicity, reproductive toxicity, and neurotoxicity there is guidance on adverse effects and their biologic markers (U.S. EPA, 1998, 1996, 1991).

To account for uncertainty and variability in the derivation of a lifetime human exposure where adverse effects are not anticipated to occur, reference values are calculated by applying a series of *uncertainty factors* to the point of departure. If a point of departure cannot be derived by modeling, a no-observed-adverse-effect level or a lowest-observed-adverse-effect level is used instead. The assessment discusses scientific considerations involving several areas of variability or uncertainty.

Human variation. The assessment accounts for variation in susceptibility across the human population and the possibility that the available data may not be representative of individuals who are most susceptible to the effect. A factor of 10 is generally used to account for this variation. This factor is reduced only if the point of departure is derived or adjusted specifically for susceptible individuals (not for a general population that includes both susceptible and non-susceptible individuals) (U.S. EPA, 2002, §4.4.5; 1998, §4.2; 1996, §4; 1994, §4.3.9.1; 1991, §3.4).

Animal-to-human extrapolation. If animal results are used to make inferences about humans, the assessment adjusts for cross-species differences. These may arise from differences in toxicokinetics or toxicodynamics. Accordingly, if the point of departure is standardized to equivalent human terms or is based on toxicokinetic or dosimetry modeling, a factor of $10^{1/2}$ (rounded to 3) is applied to account for the remaining uncertainty involving toxicokinetic and toxicodynamic differences. If a biologically based model adjusts fully for toxicokinetic and toxicodynamic differences across species, this factor is not used. In most other cases, a factor of 10 is applied (U.S. EPA, 2011;

2002, §4.4.5; 1998, §4.2; 1996, §4; 1994, §4.3.9.1; 1991, §3.4).

Adverse-effect level to no-observed-adverse-effect level. If a point of departure is based on a lowest-observed-adverse-effect level, the assessment must infer a dose where such effects are not expected. This can be a matter of great uncertainty, especially if there is no evidence available at lower doses. A factor of 10 is applied to account for the uncertainty in making this inference. A factor other than 10 may be used, depending on the magnitude and nature of the response and the shape of the dose-response curve (U.S. EPA, 2002, §4.4.5; 1998, §4.2; 1996, §4; 1994, §4.3.9.1; 1991, §3.4).

Subchronic-to-chronic exposure. If a point of departure is based on subchronic studies, the assessment considers whether lifetime exposure could have effects at lower levels of exposure. A factor of 10 is applied to account for the uncertainty in using subchronic studies to make inferences about lifetime exposure. This factor may also be applied for developmental or reproductive effects if exposure covered less than the full critical period. A factor other than 10 may be used, depending on the duration of the studies and the nature of the response (U.S. EPA, 2002, §4.4.5; 1998, §4.2; 1994, §4.3.9.1).

Incomplete database. If an incomplete database raises concern that further studies might identify a more sensitive effect, organ system, or lifestage, the assessment may apply a database uncertainty factor (U.S. EPA, 2002, §4.4.5; 1998, §4.2; 1996, §4; 1994, §4.3.9.1; 1991, §3.4). The size of the factor depends on the nature of the database deficiency. For example, the EPA typically follows the suggestion that a factor of 10 be applied if both a prenatal toxicity study and a two-generation reproduction study are missing and a factor of $10^{1/2}$ if either is missing (U.S. EPA, 2002, §4.4.5).

In this way, the assessment derives candidate values for each suitable data set and effect that is credibly associated with the agent. These results are arrayed, using common dose metrics, to show where effects occur across a range of exposures (U.S. EPA, 1994, §4.3.9).

The assessment derives or selects an *organ- or system-specific reference value* for each organ or system affected by the agent. The assessment explains the rationale for each organ/system-specific reference value (based on, for example, the highest quality studies, the most sensitive outcome, or a clustering of values). By providing these organ/system-specific reference values, IRIS assessments facilitate subsequent cumulative risk assessments that consider the combined effect of multiple agents acting at a common site or through common mechanisms (NRC, 2009).

The assessment then selects an overall reference dose and an overall reference concentration for the agent to represent lifetime human exposure levels where effects are not anticipated to occur. This is generally the most sensitive organ/system-specific reference value, though consideration of study quality and confidence in each value may lead to a different selection.

7.7. Confidence and uncertainty in the reference values

The assessment selects a standard descriptor to characterize the level of confidence in each reference value, based on the likelihood that the value would change with further testing. Confidence in reference values is based on quality of the studies used and completeness of the database, with more weight given to the latter. The level of confidence is increased for reference values based on human data supported by animal data (U.S. EPA, 1994, §4.3.9.2).

High confidence: The reference value is not likely to change with further testing,

except for mechanistic studies that might affect the interpretation of prior test results.

Medium confidence: This is a matter of judgment, between high and low confidence.

Low confidence: The reference value is especially vulnerable to change with further testing.

These criteria are consistent with guidelines for systematic reviews that evaluate the quality of evidence. These also focus on whether further research would be likely to change confidence in the estimate of effect (Guyatt et al., 2008b).

All assessments discuss the significant uncertainties encountered in the analysis. The EPA provides guidance on characterization of uncertainty (U.S. EPA, 2005a, §3.6). For example, the discussion distinguishes model uncertainty (lack of knowledge about the most appropriate experimental or analytic model) and parameter uncertainty (lack of knowledge about the parameters of a model). Assessments also discuss human variation (interpersonal differences in biologic susceptibility or in exposures that modify the effects of the agent).

August 2013

EXECUTIVE SUMMARY

Summary of Occurrence and Health Effects

Hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) is a synthetic chemical used primarily as a military explosive. RDX releases have been reported in air, water, and soil. Exposure to RDX is likely limited to individuals in or around military facilities where RDX is or was produced, used, or stored. Oral exposure may occur from drinking contaminated groundwater or ingesting crops irrigated with contaminated water. Inhalation or dermal exposures are more likely in occupational settings.

Epidemiological studies provide only limited information on worker populations exposed to RDX; several case reports describe effects primarily in the nervous system following acute exposure to RDX. Animal studies of ingested RDX demonstrate toxicity, including nervous system effects, kidney and other urogenital effects, and male reproductive effects.

Results from animal studies provide suggestive evidence of carcinogenic potential for RDX based on evidence of positive trends in liver and lung tumor incidence in experimental animals. There are no data on the carcinogenicity of RDX in humans.

Effects Other Than Cancer Observed Following Oral Exposure

Nervous system effects are a human hazard of RDX exposure. Several human case reports and animal studies provide consistent evidence of an association between RDX exposure and effects on the nervous system, including seizures or convulsions, tremors, hyperirritability, hyper-reactivity, and behavioral changes. Mechanistic data support the hypothesis that RDX-induced hyperactivity and seizures likely result from inhibition of GABAergic signaling in the limbic system.

Kidney and other urogenital effects are a potential human hazard of RDX exposure based on observations in 2-year oral toxicity studies of increased relative kidney weights in male and female mice and histopathological changes in the urogenital system of male rats exposed to RDX. An increased incidence of suppurative prostatitis was identified, and is considered a marker for RDX-related urogenital effects. There is no established mode of action (MOA) for RDX-related effects on the urogenital system.

There is suggestive evidence of male reproductive effects associated with RDX exposure based on the finding of testicular degeneration in male mice exposed to RDX in the diet for 2 years, in the only mouse study conducted of that duration. There is no known MOA for male reproductive effects of RDX exposure. Evidence for effects on other organs/systems, including the liver and developmental effects, was more limited than for the endpoints summarized above.

Oral Reference Dose (RfD) for Effects Other Than Cancer

Organ-specific RfDs were derived for hazards associated with RDX exposure (see Table ES-1). These organ- or system-specific reference values may be useful for subsequent cumulative risk assessments that consider the combined effect of multiple agents acting at a common site.

Table ES-1. Organ/system-specific RfDs and overall RfD for RDX

Effect	Basis	RfD (mg/kg-d)	Study exposure description	Confidence
Nervous system	Convulsions	3×10^{-3}	Subchronic	Medium
Kidney/urogenital	Suppurative prostatitis	8×10^{-3}	Chronic	Low
Male reproductive	Testicular degeneration	8×10^{-2}	Chronic	Low
Overall RfD	**Nervous system effects**	**3×10^{-3}**	**Subchronic**	**Medium**

The overall RfD (see Table ES-2) is derived to be protective of all types of hazards associated with RDX exposure. The effect of RDX on the nervous system was chosen as the basis for the overall RfD because nervous system effects were observed most consistently across studies, species, and exposure durations, and because they represent the most sensitive human hazard of RDX exposure. Incidence of seizures or convulsions as reported in a subchronic gavage study (Crouse et al., 2006) was selected for derivation of the overall RfD as this endpoint was measured in a study that was well-conducted, utilized a test material of higher purity than other studies, and had five closely-spaced dose groups that allowed characterization of the dose-response curve. Benchmark dose (BMD) modeling was utilized to derive the point of departure (POD) for RfD derivation (expressed as the $BMDL_{01}$). A 1% response level was chosen because of the severity of the endpoint. Further, the doses associated with nervous system effects in the Crouse et al. (2006) study also caused increased mortality in the animals. Experimental animal studies provide some evidence of an association between RDX-induced convulsions and mortality. In three studies in rats (Crouse et al., 2006; Levine et al., 1983b; Cholakis et al., 1980), investigators noted that early deaths were frequently preceded by neurotoxic signs such as tremors and convulsions; however, the recorded data from Crouse et al. (2006) do not show as clear a correspondence between convulsions (and other neurotoxic signs) and mortality (Section 1.2.1).

A physiologically-based pharmacokinetic (PBPK) model was used to extrapolate the $BMDL_{01}$ derived from a rat study to a human equivalent dose (HED) based on RDX arterial blood concentration, which was then used for RfD derivation.

The overall RfD was calculated by dividing the $BMDL_{01\text{-}HED}$ for nervous system effects by a composite uncertainty factor (UF) of 100 to account for extrapolation from animals to humans (3), interindividual differences in human susceptibility (10), and uncertainty in the database (3).

Table ES-2. Summary of reference dose (RfD) derivation

Critical effect	Point of departure[a]	UF	Chronic RfD
Nervous system effects (convulsions) 90-d F344 rat study Crouse et al. (2006)	$BMDL_{01\text{-}HED}$: 0.28 mg/kg-d	100	3×10^{-3} mg/kg-d

[a]A benchmark response (BMR) of 1% was used to derive the BMD and BMDL given the severity of the endpoint. The resulting POD was converted to a $BMDL_{01\text{-}HED}$ using a PBPK model based on modeled arterial blood concentration. The concentration was derived from the area under the curve (AUC) of modeled RDX concentration in arterial blood, which reflects the average blood RDX concentration for the exposure duration normalized to 24 hours.

Effects Other Than Cancer Observed Following Inhalation Exposure

No studies were identified that provided useful information on the effects observed following inhalation exposure to RDX. Of the available human epidemiological studies of RDX, none provided data that could be used for dose-response analysis of inhalation exposures. The single experimental animal study involving inhalation exposure is not publicly available, and was excluded from consideration due to significant study limitations, including small numbers of animals tested, lack of controls, and incomplete reporting of exposure levels. Therefore, the available health effects literature does not support the identification of hazards following inhalation exposure to RDX.

Inhalation Reference Concentration (RfC) for Effects Other Than Cancer

An RfC for RDX could not be derived based on the available health effects data. While inhalation absorption of RDX particulates is a plausible route of exposure, there are no toxicokinetic studies of RDX inhalation absorption to support an inhalation model. Therefore, a PBPK model for inhaled RDX was not developed to support route-to-route extrapolation of an RfC from the RfD.

Evidence for Human Carcinogenicity

Under EPA's cancer guidelines (U.S. EPA, 2005a), there is *suggestive evidence of carcinogenic potential* for RDX. RDX induced benign and malignant tumors in the liver and lungs of mice (Parker et al., 2006; Lish et al., 1984) or rats (Levine et al., 1983b) following long-term administration in the diet. The potential for carcinogenicity applies to all routes of human exposure.

Quantitative Estimate of Carcinogenic Risk from Oral Exposure

A quantitative estimate of carcinogenic risk from oral exposure to RDX was based on the increased incidence of hepatocellular adenomas or carcinomas and alveolar/bronchiolar adenomas or carcinomas in female $B6C3F_1$ mice observed in the carcinogenicity bioassay in mice (Lish et al., 1984). This 2-year dietary study included four dose groups and a control group, adequate numbers of animals per dose group (85/sex/group, with interim sacrifices of 10/sex/group at 6 and

12 months), and detailed reporting of methods and results (including individual animal data). The initial high dose (175 mg/kg-day) was reduced to 100 mg/kg-day at week 11 due to high mortality.

Considering these data along with the uncertainty associated with the suggestive nature of the weight of the evidence for RDX carcinogenicity, quantitative analysis of the mouse tumor data may be useful for providing a sense of the magnitude of potential carcinogenic risk.

An oral slope factor (OSF) that considered the combination of female mouse liver and lung tumors was derived from BMD and BMDL estimates that correspond to a 10% extra risk (ER) of either tumor. The $BMDL_{10}$ so derived was extrapolated to the HED using $BW^{3/4}$ scaling, and an OSF was derived by linear extrapolation from the $BMDL_{10\text{-HED}}$. The OSF is 0.04 per mg/kg-day, based on the liver and lung tumor response in female mice (Lish et al., 1984).

Quantitative Estimate of Carcinogenic Risk from Inhalation Exposure

An inhalation unit risk (IUR) value was not calculated because inhalation carcinogenicity data for RDX are not available. While inhalation absorption of RDX particulates is a plausible route of exposure, there are no toxicokinetic studies of RDX inhalation absorption to support an inhalation model. Therefore, a PBPK model for inhaled RDX was not developed to support route-to-route extrapolation of an IUR from the OSF. Thus, a quantitative cancer assessment was not conducted.

Susceptible Populations and Lifestages for Cancer and Noncancer Outcomes

Little information is available on populations that may be especially vulnerable to the toxic effects of RDX. Lifestage, and in particular childhood, susceptibility has not been observed in human or animal studies of RDX toxicity. In rats, transfer of RDX from the dam to the fetus during gestation and to pups via maternal milk has been reported; however, reproductive and developmental toxicity studies did not identify effects in offspring at doses below those that also caused maternal toxicity. Data to suggest that males may be more susceptible than females to noncancer toxicity associated with RDX exposure are limited. Specifically, urogenital effects have been noted at lower doses in males than in females. Data on the incidence of convulsions and mortality provide some indication that pregnant animals may be a susceptible population, although the evidence is unclear. Some evidence suggests that cytochrome P450 (CYP450) enzymes may be involved in the metabolism of RDX, indicating a potential for genetic polymorphisms in these metabolic enzymes to affect susceptibility to RDX. Similarly, individuals with epilepsy or other seizure syndromes that have their basis in genetic mutation to $GABA_A$ receptors may represent another group that may be susceptible to RDX exposure; however, there is no information to indicate how genetic polymorphisms may affect susceptibility to RDX.

Key Issues Addressed in Assessment

Selection of a 1% benchmark response (BMR) for convulsions. In most instances, the spectrum of effects associated with chemical exposure will range in severity, with relatively less

severe effects generally occurring at doses lower than those associated with more severe or "frank" toxicity. Convulsions in rats were selected as the basis for derivation of the RDX RfD; less severe nervous system effects were generally not observed at lower doses. U.S. EPA (2012b) recommends considering the statistical and biological characteristics of the dataset when selecting a BMR, including the severity of the effect. For convulsions, a BMR level of 1% ER was selected for modeling, balancing the quantitative limitations of the available animal bioassays and the severity of this effect. Modeling convulsion incidence from Crouse et al. (2006) using this BMR resulted in a moderate extrapolation of the BMD (3.0 mg/kg-day) below the range of experimental data (dose range from Crouse et al. (2006): 4–15 mg/kg-day).

Influence of the method of oral dosing (diet and gavage). Some uncertainty in the RfD is also associated with the influence of the method of oral dosing on the magnitude of dose required to induce nervous system effects. Findings from animal studies suggest that gavage administration generally induced convulsions in experimental animals at lower doses than did dietary administration, possibly due to the bolus dose received from gavage administration resulting in a comparatively faster absorption and higher peak blood concentrations of RDX (see Section 1.2.1). The difference in neurotoxic response associated with gavage versus dietary administration is in part reflected in the 14-fold difference in the candidate POD_{HED} values derived from the Crouse et al. (2006) (gavage administration) and Levine et al. (1983b) (dietary administration) studies (see Table 2-2). A more rigorous examination of the effect of oral dosing method cannot be performed because of the differences across studies in test materials and experimental designs (e.g., test article purity and particle size, number and spacing of dose groups, exposure duration, frequency of clinical observations, and thoroughness of the reporting of observations) that could also have contributed to differences in response. As dietary administration is more representative of potential human exposures to RDX, the use of toxicity data from a gavage (bolus dosing) study may introduce uncertainty in the RfD.

Suppurative prostatitis as a marker for kidney and other urogenital effects. The candidate RfD for kidney and other urogenital effects is based on a dose-related increase in the incidence of suppurative prostatitis from a 2-year feeding study in male F344 rats (Levine et al., 1983b). This study is the only 2-year study in rats that examined the prostate. Some reports have hypothesized that the observed suppurative prostatitis is a secondary effect from a bacterial infection unrelated to RDX toxicity (ATSDR, 2012; Sweeney et al., 2012a; Crouse et al., 2006). While an opportunistic bacterial infection may have been the proximal cause of the suppurative prostatitis, the infection was considered secondary to urogenital effects associated with RDX exposure. Histopathological findings for the bladder are not definitive because the design of the principal study called for histopathological examination of the bladder only if gross abnormalities were observed. Although the pathogenesis of kidney and urogenital effects is unclear, suppurative prostatitis was considered to be a marker for the broader array of kidney and other urogenital effects observed by Levine et al. (1983b).

LITERATURE SEARCH STRATEGY | STUDY SELECTION AND EVALUATION

Literature Search and Screening Strategy

A literature search and screening strategy was applied to identify literature related to characterizing the health effects of hexadydro-1,3,5-trinitro-1,3,5-triazine (RDX). This strategy consisted of a search of online scientific databases and other sources, casting a wide net in order to identify all potentially pertinent studies. In subsequent steps, references were screened to exclude papers not pertinent to an assessment of the chronic health effects of RDX, and the remaining references were sorted into categories for further evaluation.

The literature search for RDX was conducted in four online scientific databases—PubMed, Toxline, Toxcenter, and Toxic Substances Control Act Test Submissions (TSCATS). The initial search was performed in April 2012, and literature search updates were conducted in February 2013, January 2014, and January 2015. Searches of TSCATS were performed in February 2013 and January 2015 only. The detailed search approach for these databases, including the query strings, and the numbers of citations identified per database are provided in Appendix B, Table B-1. The Department of Defense has conducted several unpublished toxicological studies on RDX; to ensure that all such studies were located, the Defense Technical Information Center (DTIC) database, a central online repository of defense-related scientific and technical information within the Department of Defense, was also searched. A separate strategy was applied in searching DTIC because of limitations in the classification and distribution of materials in DTIC; the detailed search strategy is described in Appendix B, Table B-2. Searches of the five online databases identified 1,143 citations (after electronically eliminating duplicates). The computerized database searches were supplemented by reviewing online regulatory sources, performing "forward" and "backward" searches of Web of Science (see Appendix B, Table B-3), and adding additional references that were identified during the development of the Toxicological Review (including submissions from the Department of Defense); 34 citations were obtained using these additional search strategies. In total, 1,177 citations were identified using online scientific databases and additional search strategies.

The U.S. Environmental Protection Agency (EPA) requested public submissions of additional information in 2010 (75 FR 76982; December 10, 2010). EPA also issued a request to the public for additional information in a Federal Register Notice in 2013 (78 FR 48674; August 9, 2013), and established a docket for public comment (EPA-HQ-ORD-2013-0430; available at www.regulations.gov) maintained through the development of the assessment. No submissions were received in response to these calls for data.

The citations identified using the search strategy described above were screened using the title, abstract, and in limited instances, full text for pertinence to examining the health effects of chronic RDX exposure. The process for screening the literature is described below and is shown graphically in Figure LS-1.[1] The objective of this manual screen was to identify sources of primary human health effects data and sources of primary data that inform the assessment of RDX health effects (i.e., the bottom three boxes in Figure LS-1). Inclusion and exclusion criteria used to manually screen the references in order to identify health effect studies (i.e., the green boxes in Figure LS-1) are provided in Table LS-1. Specific inclusion criteria were not applied in identifying sources of mechanistic and toxicokinetic data. The number of such studies for RDX is not large, and therefore, all studies that provided data on adsorption, distribution, metabolism, or elimination, physiologically-based pharmacokinetic (PBPK) models, or relevant RDX mode of action (MOA) were considered. Studies that met one or more of the exclusion criteria in Table LS-1 were binned as "Excluded/Not on Topic" and were not further considered in this assessment. A final group of studies consisted of reviews and other sources of RDX information (e.g., exposure, ecosystem effects) that did not meet the inclusion criteria in Table LS-1. These studies were binned into a category called "Secondary Literature and Sources of Other RDX Information," and were considered as appropriate during development of this assessment.

The results of this literature screening are described below and graphically in Figure LS-1:

- 25 references (including both human and animal studies) were identified as sources of health effects data and were considered for data extraction to evidence tables and exposure-response arrays.[2]

- 25 references were identified as sources of supplementary health effects data, including experimental animal studies involving acute or short-term exposures or dermal exposure, and human case reports. Studies investigating the effects of acute/short-term and dermal exposures and case reports are generally less pertinent for characterizing health hazards associated with chronic oral and inhalation exposure. Therefore, information from these studies was not extracted into evidence tables. Nevertheless, these studies were still considered as possible sources of supplementary health effects information.

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[1]Studies were assigned (or "tagged") to a given category in Health and Environmental Research Online (HERO) that best reflected the primary content of the study. In general, studies were not assigned multiple tags in order to simplify the tracking of references. Nevertheless, the inclusion of a citation in a given category (or tag) did not preclude its use in one or more other categories. For example, Woody et al. (1986), a case report of accidental ingestion of RDX by a child, was tagged to the human case reports under Supplementary Studies in Figure LS-1. This case report also provides pharmacokinetic data and was a pertinent source of information on RDX toxicokinetics, but was not assigned a second tag for toxicokinetics.

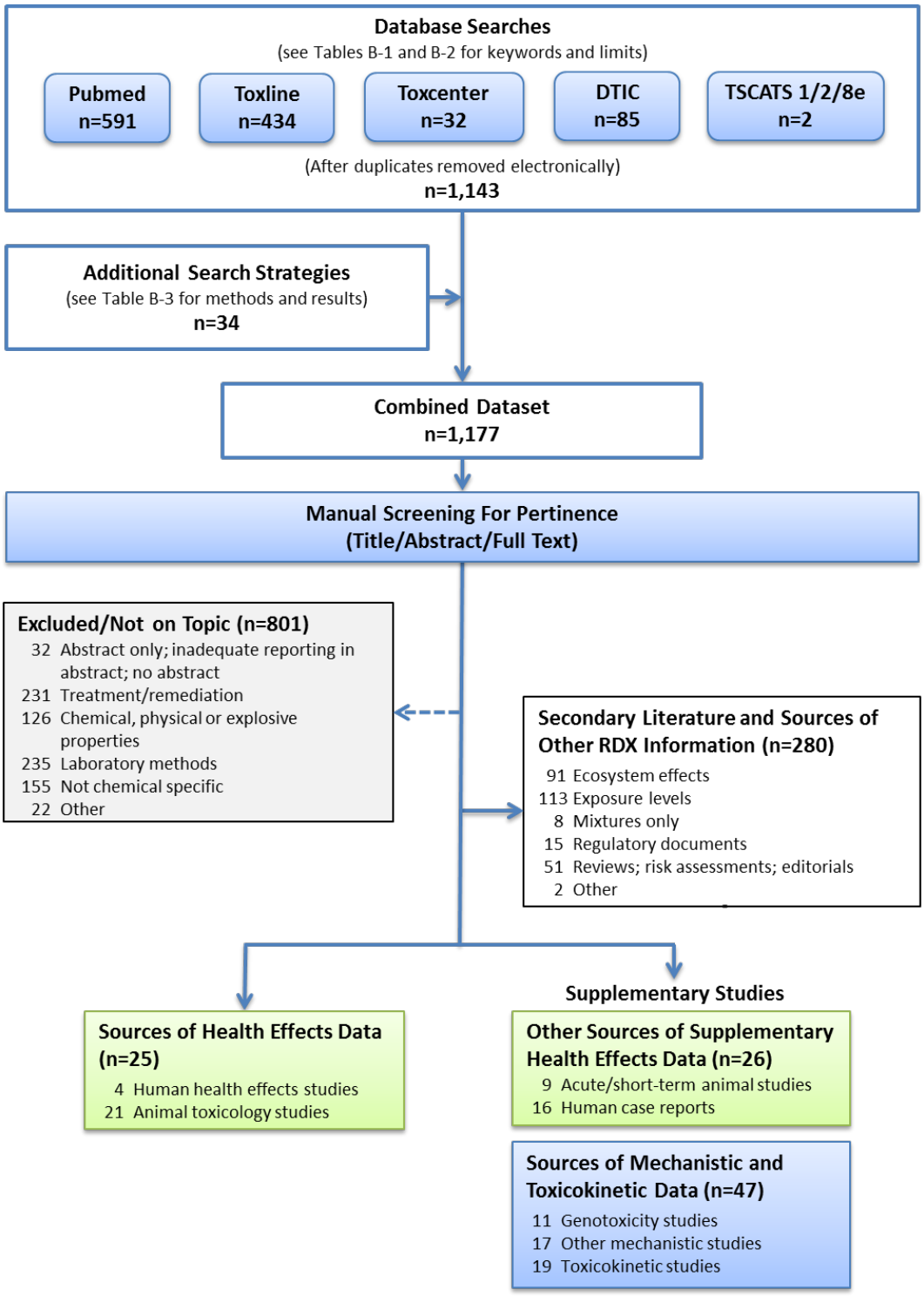
[2] On the HERO project page, 27 records are associated with "Sources of Health Effects Data," rather than the 25 identified in Figure LS-1. Two of the records in HERO are not unique studies or reports. Rather, these two records provide links to the multi-volume laboratory reports for the 2-year studies in rats by Levine et al. (1983b) and mice by Lish et al. (1984)

- 47 references were identified as sources of mechanistic and toxicokinetic data; these included 19 studies describing PBPK models and other toxicokinetic information, 11 studies providing genotoxicity information, and 17 studies pertaining to other mechanistic information. Information from these studies was not extracted into evidence tables; however, these studies supplemented the assessment of RDX health effects (e.g., evaluation of MOA and extrapolation of experimental animal findings to humans).

- 280 references were identified as secondary literature (e.g., reviews and other agency assessments) or as studies providing potentially useful information on RDX (e.g., studies providing information on exposure levels or effects on nonmammalian species); these references were kept as additional resources for development of the Toxicological Review.

- 801 references were identified as not being pertinent (or not on topic) to an evaluation of the chronic health effects of RDX and were excluded from further consideration (see Figure LS-1 and Table LS-1 for exclusion criteria). Retrieving a large number of references that are not on topic is a consequence of applying an initial search strategy designed to cast a wide net and to minimize the possibility of missing potentially relevant health effects data.

1



The numbers on this figure match the HERO project page as of 1/12/2016; subsequent changes may not be reflected. A limited number of references were assigned more than one tag; therefore, the sum of the references in boxes below "Manual Screening for Pertinence" does not match exactly the total number of references in the "Combined Dataset."

Figure LS-1. Summary of literature search and screening process for RDX.

Table LS-1. Inclusion-exclusion criteria for health effect studies

	Inclusion criteria	Exclusion criteria
Population	• Humans • Standard mammalian animal models, including rat, mouse, rabbit, guinea pig, monkey, dog	• Ecological species[a] • Nonmammalian species[a]
Exposure	• Exposure is to RDX • Exposure is measured in an environmental medium (e.g., air, water, diet) • Exposure via oral or inhalation routes	• Study population is not exposed to RDX • Exposure to a mixture only • Exposure via injection (e.g., intravenous [i.v.])
Outcome	• Study includes a measure of one or more health effect endpoints, including effects on the nervous, kidney/urogenital, musculoskeletal, cardiovascular, immune, and gastrointestinal systems, reproduction, development, liver, eyes, and cancer	
Other		Not on topic, including: • Abstract only, inadequately reported abstract, or no abstract, and not considered further because study was not potentially relevant • Bioremediation, biodegradation, or chemical or physical treatment of RDX and other munitions, including evaluation of wastewater treatment technologies and methods for remediation of contaminated water and soil • Chemical, physical, or explosive properties, including studies of RDX crystal quality, energetics characteristics, sublimation kinetics, isotope ratios, and thermal decomposition and other explosive properties • Analytical methods for measuring/detecting/ remotely sensing RDX in environmental media, and use in sample preparations and assays • Not chemical specific (studies that do not involve testing of RDX) • Other studies not informative for evaluating RDX health effects and not captured by other exclusion criteria, including: -- Superfund site records of decision that describe remedial action plans for waste sites -- characterization of waste sites contaminated by explosives -- foreign language studies where translation was not warranted because, based on title or abstract, the added value to the evaluation of RDX health effects

	Inclusion criteria	Exclusion criteria
		was considered small (e.g., Chinese paper of case reports of RDX poisonings) -- duplicate studies not previously identified

[a]Studies that met this exclusion criterion were not considered a source of health effects or supplementary health effects data, but were considered as other sources of information potentially useful in assessing the health effects of RDX.

The documentation and results for the literature search and screen can be found on the Health and Environmental Research Online (HERO) website on the RDX project page at: (http://hero.epa.gov/index.cfm/project/page/project_id/2216).

Selection of Critical Studies for Presentation in Evidence Tables

Selection of Critical Studies

In order to systematically summarize the important information from the primary health effects studies in the RDX database, evidence tables were constructed in a standardized tabular format as recommended by the NRC (2011). Of the studies that were retained after the literature search and screen, 25 were categorized as "Sources of Health Effects Data" (Figure LS-1, Table LS-1) and were considered for extraction into evidence tables for hazard identification in Chapter 1.

A study was not presented in the evidence tables if flaws in its design, conduct, or reporting were so great that the results would not be considered credible (e.g., studies where concurrent control information is lacking). Such study design flaws are discussed in a number of EPA's guidelines (see http://www.epa.gov/iris/backgrd.html) or summarized in the Preamble. For RDX, four studies were considered uninformative and were removed from further consideration in the assessment because of fundamental issues with study design, conduct, or reporting. The specific studies and basis for considering the studies to be uninformative are summarized in Table LS-2.

Table LS-2. Studies determined not to be informative because of significant issues with design, conduct, or reporting

Reference	Rationale for exclusion
Haskell Laboratories (1942); 14-wk study in dogs	Incomplete information on exposure levels; breed of dog was not reported; inadequate reporting of results; sections of document were illegible.
von Oettingen et al. (1949); 10-wk oral study in rats	No control group; strain of rat was not reported.
ATSDR (1996); Disease prevalence study in residential population	Study of a population residing in two neighborhoods where RDX had been detected in well water. The study was conducted 7 yrs after residents were provided the opportunity to connect to a municipal water supply. Only one target-area household reported using private well water for bathing and cooking at the time of the health study. The study was not considered informative because the design was not able to adequately define the exposed population.
Unpublished report (dated 1944) from the DTIC database; Human and animal data	One section of the report describes a human case series with no referent group. Issues with the inhalation experimental animal studies included lack of control groups, incomplete information on exposure levels, and inadequate reporting of results. [Because this report is classified as a limited distribution document in the DTIC database, it was not added to the HERO project page for RDX.]

The health effects literature for RDX is not extensive. With the exception of the studies listed in Table LS-2 (i.e., those determined to be uninformative), all human and experimental animal studies of RDX involving repeated exposure were considered in assessing the evidence for health effects associated with chronic exposure to RDX.

Studies that contain pertinent information for the toxicological review and augment hazard identification conclusions, such as genotoxicity and other mechanistic studies, studies describing the toxicokinetics of RDX, human case reports, and experimental animal studies involving exposures of acute/short-term duration or routes of exposure other than oral and inhalation, were not included in evidence tables. Nevertheless, these studies were considered, where relevant, in the evaluation of RDX health hazards.

Study Evaluation

For this assessment, primary sources of health effects data consisted of three human studies[3] and 21 reports[4] presenting results of experimental animal studies. These studies were evaluated using the study quality considerations outlined in the Preamble, considering aspects of design, conduct, or reporting that could affect the interpretation of results, overall contribution to the synthesis of evidence, and determination of hazard potential as noted in various EPA guidance documents (U.S. EPA, 2005a, 2002, 1994). The objective was to identify the stronger, more informative studies based on a uniform evaluation of quality characteristics across studies of similar design.

Additionally, a number of general questions, presented in Table LS-3, were considered in evaluating the animal studies. Much of the key information for conducting this evaluation can be determined based on study methods and how the study results were reported. Importantly, the evaluation at this stage does not consider the direction or magnitude of any reported effects.

Table LS-3. Considerations and relevant experimental information for evaluation of experimental animal studies

Methodological feature	Considerations (relevant information extracted into evidence tables)
Test animal	Suitability of the species, strain, sex, and source of the test animals
Experimental design	Suitability of animal age/lifestage at exposure and endpoint testing; periodicity and duration of exposure (e.g., hrs/d, d/wk); timing of endpoint evaluations; and sample size and experimental unit (e.g., animals, dams, litters)
Exposure	Characterization of test article source, composition, purity, and stability; suitability of the control (e.g., vehicle control); documentation of exposure techniques (e.g., route, chamber type, gavage volume); verification of exposure levels (e.g., consideration of homogeneity, stability, analytical methods)
Endpoint evaluation	Suitability of specific methods for assessing the endpoint(s) of interest
Results presentation	Data presentation for endpoint(s) of interest (including measures of variability) and for other relevant endpoints needed for results interpretation (e.g., maternal toxicity, decrements in body weight in relation to organ weight)

[3]Two reports with human data were determined not to be informative; see Table LS-2. The study by ATSDR (1996) was included in HERO and in Figure LS-1. The unpublished report from the DTIC database was not included in either HERO or Figure LS-1 because this report is classified as a limited distribution document in DTIC. This accounts for the three human studies being reviewed for study evaluation rather than the four identified in the literature search (see Figure LS-1).

[4]The number of reports of experimental animal studies does not equal the number of studies. The results of some studies were documented in multiple reports (e.g., a 2-year study in F344 rats by Levine et al. (1983b) was published in three volumes). The Cholakis et al. (1980) study included, in a single report, subchronic studies in rats and mice, a 2-generation reproductive toxicity study in rats, and developmental toxicity studies in rats and rabbits.

Information on study features related to this evaluation is reported in evidence tables and was considered in the synthesis of evidence. Discussion of study strengths and limitations (that ultimately supported preferences for the studies and data relied upon) were included in the text where relevant. If EPA's interpretation of a study differed from that of the study authors, the assessment discusses the basis for the difference.

The general findings of this evaluation are presented in the remainder of this section. Study evaluation considerations that are outcome specific are discussed in the relevant health effect sections in Section 1.2.

Human Studies

The body of literature on RDX includes three studies of populations occupationally exposed to RDX (one case-control and two cross-sectional studies) (West and Stafford, 1997; Hathaway and Buck, 1977). To varying degrees, these epidemiology studies are limited in their ability to assess the relationship between RDX exposure and the incidence of human health effects. Some studies lacked information related to study design, such as a precise definition of the study population, while others did not include a comprehensive exposure assessment or details regarding potential confounders. All three studies had small sample sizes (60–69 exposed workers in the cross-sectional studies and 32 cases in the case-control study), which limits their statistical power when comparing exposed workers or cases and unexposed or control participants.

The study by Ma and Li (1993) of Chinese industrial workers provided limited information on participant recruitment, selection, and participation rate; the available information was not adequate to evaluate the potential for selection bias. Also, no information on adjustment for co-exposure to trinitrotoluene (TNT) or other neurological risk factors (e.g., alcohol consumption) was provided. The study by Hathaway and Buck (1977) included details on exposure assessment, but did not provide information on length of employment or other metrics that could be used to ascertain duration of exposure. In the case-control study by West and Stafford (1997), RDX was identified as one of the many chemicals that workers may have been exposed to in the ordnance factory. Thus, there is a potential for co-exposure to other chemicals that may elicit the observed effects. The methodological limitations in these three studies were considered in the synthesis of evidence for each of the health effects and in reaching determinations of hazard (see Section 1.2).

In addition to the aforementioned studies, the human health effects literature includes 16 case reports that describe effects following acute exposure to RDX. Case reports can suggest organ systems and health outcomes that might be related to RDX exposure but are often anecdotal, and typically describe unusual or extreme exposure situations; thus, they provide little information that would be useful for characterizing chronic health effects. Therefore, RDX case reports were only briefly reviewed; a critical evaluation was not undertaken. A summary of these case reports is provided in Appendix C, Section C.2.

Experimental Animal Studies

The oral toxicity database for RDX includes three chronic studies in rats and mice, eight subchronic studies in rats, mice, dogs, and monkeys, two shorter-term studies in dogs and rats, one two-generation reproductive toxicity study in the rat, four developmental toxicity studies in rats and rabbits, and a single-exposure study of audiogenic seizures in rats (Table LS-4).

Table LS-4. Summary of experimental animal database

Study category	Study duration, species/strain, and oral administration method
Chronic	2-Yr study in B6C3F$_1$ mice (diet) (Lish et al., 1984) 2-Yr study in Sprague-Dawley rats (diet) (Hart, 1976) 2-Yr study in F344 rats (diet) (Levine et al., 1983b)
Subchronic	13-Wk study in B6C3F$_1$ mice, experiment 1 (diet) (Cholakis et al., 1980) 13-Wk study in B6C3F$_1$ mice, experiment 2 (diet) (Cholakis et al., 1980) 13-Wk study in F344 rats (diet) (Cholakis et al., 1980) 13-Wk study in F344 rats (diet) (Levine et al., 1990; Levine et al., 1981a, b) 13-Wk study in F344 rats (gavage) (Crouse et al., 2006) 13-Wk study in rats, strain not specified (diet) (von Oettingen et al., 1949) 13-Wk study in beagle dogs (diet) (Hart, 1974) 13-Wk study in monkeys (gavage) (Martin and Hart, 1974) 6-Wk study in dogs, breed not specified (diet) (von Oettingen et al., 1949) 30-D study in Sprague-Dawley rats (gavage) (MacPhail et al., 1985)
Reproductive	2-Generation reproductive toxicity study in CD rats (diet) (Cholakis et al., 1980)
Developmental	Developmental study (gestational days [GDs] 6–19) in F344 rats (gavage) (Cholakis et al., 1980) Developmental study (GDs 6–15) in Sprague-Dawley rats, range-finding (gavage) (Angerhofer et al., 1986) Developmental study (GDs 6–15) in Sprague-Dawley rats (gavage) (Angerhofer et al., 1986) Developmental study (GDs 7–29) in New Zealand White (NZW) rabbits (gavage) (Cholakis et al., 1980)
Nervous system	8-Hr study of audiogenic seizures in Long Evans rats (gavage) (Burdette et al., 1988)

With the exception of two studies (Levine et al., 1990; von Oettingen et al., 1949), these toxicity studies are available only as unpublished contract laboratory reports. Peer reviews of three unpublished studies identified as most informative to the assessment of the health effects of RDX—the 2-year bioassays by Levine et al. (1983b) and Lish et al. (1984) and the subchronic toxicity study by Crouse et al. (2006)—were conducted by Versar, Inc. for EPA in 2012. The report of the peer reviews (U.S. EPA, 2012d) is available at https://epa.gov/hero. The peer reviewers generally concluded that the 2-year bioassay reports provided useful information on the toxicity of RDX, noting that there were limitations in interpretation due to aspects of the histopathological analysis and the statistical approaches employed. The peer reviewers similarly determined that the report by Crouse et al. (2006) provided useful information on RDX toxicity, including an array of endpoints for neurotoxicity and immunotoxicity (U.S. EPA, 2012d).

Only one unpublished inhalation study of RDX (dated 1944) was identified. As discussed in Appendix B and Table LS-2, this inhalation study was considered uninformative and was excluded

from consideration in the development of the Toxicological Review because of study design issues (including lack of a control group, incomplete information on exposure levels, and inadequate reporting). Therefore, evaluation of the experimental animal database for RDX is limited to studies of oral toxicity. An evaluation of the oral toxicity literature, organized by general methodological features, is provided in the remainder of this section.

Test animal

The RDX database consists of health effect studies conducted in multiple strains of rats (F344, Sprague-Dawley, CD), mice ($B6C3F_1$), dogs (beagle), and monkeys. The species and strains of animals used are consistent with those typically used in laboratory studies. All of these species or strains were considered relevant to assessing the potential human health effects of RDX. Several studies in the RDX database provided inadequate information on test animals. The strain of monkey (Rhesus or Cynomolgus) used in the study by Martin and Hart (1974) was not clearly specified. In one study, the breed of dog and strain of rat were unreported (von Oettingen et al., 1949). The species, strain, and sex of the animals used are recorded in the evidence tables.

Other studies of RDX were identified that used nonstandard species, including deer mice (*Peromyscus maniculatus*), western fence lizards (*Sceloporus occidentalis*), prairie voles (*Microtus ochrogaster*), and northern bobwhite quail (*Colinus virginianus*). These studies provide information relevant to RDX toxicokinetics and mechanism of action on the nervous system, but not health effects data. Therefore, these studies are not included in evidence tables, but are discussed where relevant in the assessment.

Experimental setup

General aspects of study design and experimental setup were evaluated for all studies that included health effects data to determine if they were appropriate for evaluation of specific endpoints. Key features of the experimental setup, including the periodicity and duration of exposure, timing of exposure (e.g., gestational days for developmental studies), experimental group sample sizes, and interim sacrifices are summarized in the evidence tables. Note that sample size was not a basis for excluding a study from consideration, as studies with a small number of animals can still inform the consistency of effects observed for a specific endpoint. Nevertheless, the informativeness of studies with small sample sizes, e.g., three animals/sex/group in the case of Hart (1974) and Martin and Hart (1974), was reduced. Elements of the experimental setup that could influence interpretation of study findings are discussed in the relevant hazard identification sections of the assessment.

Exposure

Properties of the test material were also considered in determining whether the exposures were sufficiently specific to the compound of interest. Two properties of the RDX test materials that varied across experimental animal studies and that were taken into consideration in evaluating the evidence for RDX hazards are the particle size and purity of the test material. The purity of RDX

used in health effects studies varied from 84 to 99.99%. The major contaminants were octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX) and water, which are the primary contaminants of RDX produced during manufacturing. The majority of studies used RDX with ~10% impurities; only Crouse et al. (2006) used 99.99% pure RDX as a test material in their study. The toxicity of HMX was assessed by the Integrated Risk Information System (IRIS) Program in 1988 (http://cfpub.epa.gov/ncea/iris2/chemicalLanding.cfm?substance_nmbr=311); histopathological changes in the liver in male F344 rats and in the kidney in female rats were reported in a 13-week feeding study. No chronic studies were available to evaluate the carcinogenicity of HMX. The presence of the impurities introduces some uncertainty in attribution of toxicity to RDX. However, consistency in the doses at which some toxic effects were seen across studies suggests that the uncertainty associated with the use of less pure test materials may be relatively small. Evidence of neurotoxic effects in the study with 99.99% pure RDX occurred at doses of 8–15 mg/kg-day; studies with less pure RDX reported similar symptoms at doses ≥20 mg/kg-day. It should be noted that the test materials employed in these studies (i.e., with ~10% impurities) are consistent with the purity of RDX that would be released into the environment.

Differences in milling procedures used to generate the test material resulted in the use of RDX of varying particle sizes across studies. Some studies utilized a test material with a relatively fine particle size (majority of particles <66 μm in size), while others used a test material with comparatively coarse particle size (~200 μm particle size). Differences in particle size across studies could result in different rates of absorption of RDX into the blood stream, which could account for differences in response observed across studies, including neurotoxicity. Information on test material purity and particle size, as provided by study authors, is reported in the evidence tables, and was considered in evaluating the toxicity of RDX. The lack of characterization of the test material in the studies by Hart (1974), Hart (1976), and Martin and Hart (1974) was considered a deficiency.

Endpoint evaluation procedures

Some methodological considerations used to evaluate studies of RDX toxicity are outcome specific—in particular, effects on the nervous system and development. Outcome-specific methodological considerations are discussed in the relevant health effect sections in Section 1.2. For example, many of the studies that noted neurotoxicity in the form of seizures or convulsions were not designed to assess that specific endpoint and reported the number of animals with seizures as part of clinical observations that, in general, were recorded only once daily. This frequency of observations could have missed neurobehavioral events. While these studies can provide qualitative evidence of neurotoxicity, they may have underestimated the true incidence of seizures or convulsions because they were not designed to systematically evaluate neurotoxic outcomes.

Outcomes and data reporting

In evaluating studies, consideration was given to whether data were reported for all endpoints specified in the methods section and for all study groups, and whether any data were excluded from presentation or analysis. For example, it was noted where histopathological analysis was limited to control and high-dose groups, a study reporting feature that limited the ability to identify dose-related trends. In limited cases, EPA performed additional statistical analysis to identify trends or refine analyses consistent with EPA guidance (e.g., analyzing developmental data sets on a per litter basis rather than by individual fetus). Study results have been extracted and presented in evidence tables.

Notable features of the RDX database

Three 2-year toxicity bioassays of RDX are available as unpublished laboratory studies (Lish et al., 1984; Levine et al., 1983b; Hart, 1976). The bioassays by Levine et al. (1983b) in the rat and by Lish et al. (1984) in the mouse were conducted in accordance with Food and Drug Administration (FDA) Good Laboratory Practices (GLPs) in place at the time of the studies. Both studies included interim sacrifices (at 6 and 12 months). Complete histopathological examinations were performed on all animals in the control and high-dose groups; however, only a subset of tissues was examined in the mid-dose groups, limiting the ability to identify dose-related trends for tissues with incomplete histopathology. Additionally, in the mouse bioassay by Lish et al. (1984), the initial high dose (175 mg/kg-day) was reduced to 100 mg/kg-day at week 11 because of high mortality, thereby reducing the number of high-dose animals on study for the full 2 years of dosing (see Table LS-5).

An earlier unpublished 2-year study in rats by Hart (1976) used a dose range that was lower than the Levine et al. (1983b) and Lish et al. (1984) bioassays. Histopathology findings were limited by the lack of pathology examinations in the mid-dose groups and lack of individual time of death, which impacts the ability to interpret the histopathology data. In addition, a heating system malfunction on days 75 - 76 of the study resulted in the death of 59 rats from the control and treatment groups, thereby reducing the number of animals in the study (see Table LS-5).

Experimental animal toxicity studies of RDX involving less-than-lifetime exposure (Crouse et al., 2006; Angerhofer et al., 1986; MacPhail et al., 1985; Levine et al., 1981a; Cholakis et al., 1980; Hart, 1974; Martin and Hart, 1974; von Oettingen et al., 1949) were published or reported between the years 1949 and 2006, and differences in robustness of study design, conduct, and reporting reflect that time span. All but two of the eight short-term and subchronic toxicity studies of RDX are available as unpublished laboratory studies; published studies include von Oettingen et al. (1949) and Levine et al. (1981a), a laboratory report of a 13-week study of RDX in F344 rats with subsets of the data subsequently published as Levine et al. (1981b) and Levine et al. (1990). The majority of studies conducted histopathological examinations on only some of the experimental groups (e.g., control and high dose). One subchronic study Crouse et al. (2006) was peer-reviewed by Versar, Inc. for EPA in 2012. The peer reviewers determined that the report provided useful

information on the toxicity of RDX, including an array of endpoints for neurotoxicity and immunotoxicity (U.S. EPA, 2012d). The assessment of neurotoxicity in the study could have been improved with more histological evaluation as well as additional behavioral assessment.

Some of the more important limitations in study design, conduct, and reporting of experimental animal toxicity studies of RDX are summarized in Table LS-5. Limitations of these studies as well as the study evaluation consideration described in this section were taken into consideration in evaluating and synthesizing the evidence for each of the health effects in Section 1.2.

Table LS-5. Experimental animal studies considered less informative because of certain study design, conduct, or reporting limitations

References	Study design, conduct, and reporting limitations
Lish et al. (1984) 2-yr mouse study	The initial high dose (175 mg/kg-d) was reduced to 100 mg/kg-d at wk 11 due to high mortality. Mortality of surviving mice was similar to controls after dose reduction.
Hart (1976) 2-yr rat study	A heating system malfunction on d 75–76 of the study resulted in the deaths of 59 rats from the control and treatment groups. Dead animals were subsequently eliminated from the analysis. Interpretation of the histopathology findings was limited by the lack of pathology examinations in the mid-dose groups and lack of individual time of death. Test material poorly characterized; purity was not reported.
Cholakis et al. (1980) 13-wk mouse study (Experiment 1)	The dose range was too low to produce effects in mice. Histopathological examinations were not performed.
Cholakis et al. (1980) 13-wk mouse study (Experiment 2)	Nonstandard dosing regimen followed: 0, 40, 60, or 80 mg/kg-d for 2 wks. For the next 11 wks, the dosing was inverted, so that the 40 mg/kg-d group received 320 mg/kg-d, the 60 mg/kg-d group received 160 mg/kg-d, and the 80 mg/kg-d group continued to receive the same dose. The rationale for this dosing regimen was not provided in the study report.
Levine et al. (1981a) 13-wk rat study	Analysis of one lot of rodent feed showed measurable levels of contaminants, including chlorinated pesticides (dieldrin, heptachlor epoxide, beta-hexachlorocyclohexane [BHC], and dichlorodiphenyltrichloroethane [DDT]), polychlorinated biphenyls (PCBs), and organophosphates (methyl parathion, carbophenothion, and disulfeton).
Martin and Hart (1974) 13-wk monkey study	The species of monkey is unclear (either Cynomolgus or Rhesus). Some test subjects may have had variable dosing due to emesis. Small sample size per dose group (n = 3). Test material poorly characterized; purity was not reported.
von Oettingen et al. (1949) 12-wk rat study	The strain of rat was not reported. Only gross observations were made at autopsy.
von Oettingen et al. (1949) 6-wk dog study	The breed of dog was not reported. Only gross observations were made at autopsy.

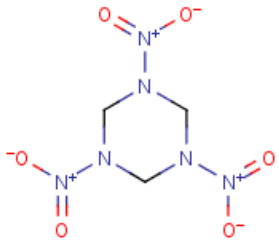
1. HAZARD IDENTIFICATION

1.1. Overview of Chemical Properties and Toxicokinetics

1.1.1. Chemical Properties

Hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) is a member of the nitramine class of organic nitrate explosives (Boileau et al., 2003; Bingham et al., 2001) and is not found naturally in the environment. It has low solubility in water (Yalkowsky and He, 2003) and slowly volatilizes from water or moist soil (ATSDR, 2012). The normalized soil organic carbon/water partition coefficient (K_{OC}) values for RDX indicate a potential for RDX to be mobile in soil (Spanggord et al., 1980a). The vapor pressure suggests that RDX will exist as particulate matter in air and be removed by both wet and dry deposition (Spanggord et al., 1980a). RDX degrades in the environment and can be subject to both photolysis (Sikka et al., 1980; Spanggord et al., 1980a) and biodegradation (Funk et al., 1993; McCormick et al., 1981) (Table 1-1).

Table 1-1. Chemical identity and physicochemical properties of RDX

Characteristic or property	Value	Reference
Chemical structure		NLM (2011)
CASRN	121-82-4	
Synonyms	Hexahydro-1,3,5-trinitro-s-triazine; 1,3,5-trinitro-1,3,5-triazacyclohexane; 1,3,5-trinitrohexahydro-s-triazine; cyclonite; cyclotrimethylenetrinitramine; hexogen; cyclotrimethylenenitramine; Research Department Explosive; Royal Demolition eXplosive; RDX	
Color/form	White, crystalline solid	Bingham et al. (2001)
Molecular formula	$C_3H_6N_6O_6$	NLM (2011)
Molecular weight	222.12	Lide (2005)
Density (g/cm^3 at 20°C)	1.82	Lide (2005)
Boiling point (°C)	276–280	Bingham et al. (2001)

Characteristic or property	Value	Reference
Melting point (°C)	205.5	Lide (2005)
Heat of formation (kJ/g)	−0.277	Ryon et al. (1984)
Log K_{OW}	0.87–0.90	Hansch et al. (1995)
K_{OC}	42–167	Spanggord et al. (1980b)
Henry's law constant (atm-m^3/mole at 25°C)	2.0×10^{-11}	ATSDR (2012)
Vapor pressure (mm Hg at 20˚C)	4.10×10^{-9}	Spanggord et al. (1980a)
Solubility in water (mg/L at 25°C)	59.7	Yalkowsky and He (2003)

1.1.2. Toxicokinetics

RDX is absorbed following exposure by oral and inhalation routes (see Appendix C, Section C.1.1). Studies in experimental animals indicate that oral absorption rates can range from approximately 50 to 90% (Krishnan et al., 2009; Guo et al., 1985; Schneider et al., 1978, 1977), with the rate and extent of absorption dependent on the physical form of RDX (i.e., the increased surface area associated with finely powdered RDX allows for increased absorption) and the dosing preparation or matrix (Bannon et al., 2009a; Krishnan et al., 2009; Crouse et al., 2008; Bannon, 2006; Guo et al., 1985; MacPhail et al., 1985; Schneider et al., 1977). Dermal absorption of RDX has been demonstrated in in vitro studies using human and pig skin (Reddy et al., 2008; Reifenrath et al., 2008).

RDX is systemically distributed, including to the brain (i.e., RDX can cross the blood:brain barrier), heart, kidney, liver, and fat (Musick et al., 2010; Bannon et al., 2006; MacPhail et al., 1985; Schneider et al., 1977). In rats, RDX can be transferred from dam to fetus across the placental:blood barrier, and has been identified in maternal milk (Hess-Ruth et al., 2007).

The metabolism of RDX in humans has not been investigated. Studies in experimental animals indicate that metabolism of RDX is extensive and includes denitration, ring cleavage, and generation of CO_2 possibly through cytochrome P450 (CYP450) (Musick et al., 2010; Major et al., 2007; Fellows et al., 2006; Bhushan et al., 2003; Schneider et al., 1978, 1977).

RDX and metabolites are eliminated primarily via urinary excretion and exhalation of CO_2 (Sweeney et al., 2012a; Musick et al., 2010; Krishnan et al., 2009; Major et al., 2007; Schneider et al., 1977). Estimated elimination half-lives ($t_{1/2}$) indicate that RDX is more rapidly metabolized in mice than in rats and humans; estimated $t_{1/2}$ values were 1.2 hours for mice, 5–10 hours for rats, and 15–29 hours for humans (Sweeney et al., 2012b; Krishnan et al., 2009; Özhan et al., 2003; Woody et al., 1986; Schneider et al., 1977).

A more detailed summary of RDX toxicokinetics is provided in Appendix C, Section C.1.

1.1.3. Description of Toxicokinetic Models

A physiologically based pharmacokinetic (PBPK) model to simulate the pharmacokinetics of RDX in rats was first developed by Krishnan et al. (2009) and revised to extend the model to humans and mice (Sweeney et al., 2012a; Sweeney et al., 2012b). The Sweeney et al. (2012a) model consists of six main compartments: blood, brain, fat, liver, and lumped compartments for rapidly perfused tissues and slowly perfused tissues, and can simulate RDX exposures via the intravenous (i.v.) or oral route. This model assumes that the distribution of RDX to tissues is flow-limited, and represents oral absorption as first-order uptake from the gastrointestinal (GI) tract into the liver, with 100% of the dose absorbed. RDX is assumed to be cleared by first-order metabolism in the liver. The model does not represent the kinetics of any RDX metabolites. The Sweeney et al. (2012a) and Sweeney et al. (2012b) PBPK models were evaluated and subsequently modified by the U.S. Environmental Protection Agency (EPA) for use in dose-response modeling in this assessment (see Appendix C, Section C.1.5).

1.2. PRESENTATION AND SYNTHESIS OF EVIDENCE BY ORGAN/SYSTEM

In experimental animal studies, RDX test material administered in toxicology studies included formulations that ranged in purity (from 84 to 99.99%) and in particle size (from <66 to ~200 μm particle size). Differences in test material purity and particle size were taken into consideration while evaluating RDX toxicity findings; this is discussed in the literature search section and incorporated in the synthesis of evidence.

Mortality has been reported in the animal toxicology studies conducted for RDX. Due to the serious nature associated with a frank effect such as mortality, EPA specifically evaluated the database with respect to mortality (see Appendix C, Section C.3.1). In brief, mortality was observed following exposure to a range of doses in chronic-duration studies, in studies up to 6 months in duration, and during gestation. In further analyzing the available evidence, mortality occurred at lower doses in rats compared with mice and following gavage administration compared with dietary administration. Additionally, mortality occurred to a greater extent with administration of RDX in the form of relatively finer particle sizes, potentially due to the reduced ability of larger particles of RDX to enter the bloodstream. Some investigators attributed the mortality to RDX-related cancer or noncancer effects (e.g., kidney or nervous system effects); others identified no cause for the animal deaths. Typically, evidence related to various hazards is presented and synthesized in distinct organ- or system-specific sections. However, in this case, the assessment does not present mortality in a hazard section by itself due to the likelihood that events leading to mortality fall under other specific hazards. Mortality evidence is considered in discussions of the evidence for organ/system-specific hazards where applicable.

1.2.1. Nervous System Effects

In humans, nervous system effects following RDX exposure have been observed in multiple case reports, and the association between RDX exposure and neurobehavioral effects has been examined in a single cross-sectional occupational epidemiology study. Information relevant to an examination of the association between RDX exposure and nervous system effects also comes from experimental animal studies involving chronic, subchronic, and gestational exposure to ingested RDX. A summary of nervous system effects associated with RDX exposure is presented in Tables 1-2 and 1-3 and Figure 1-1. Experimental animal studies are ordered in the evidence table and exposure-response array by duration of exposure and then species.

Observational Studies in Humans

In a cross-sectional study by Ma and Li (1993), neurobehavioral effects were evaluated in Chinese workers occupationally exposed to RDX. Memory retention and block design scores[5] were significantly lower among exposed workers (mean concentrations of RDX in two exposed groups: 0.407 and 0.672 mg/m^3) compared to unexposed workers from the same plant. However, no significant differences were observed between the groups on other neurobehavioral tests (e.g., simple and choice reaction times, and letter cancellation test) (Table 1-2). This study did not consider potential confounders such as alcohol consumption or co-exposure to trinitrotoluene (TNT), and there was limited information characterizing exposure to RDX.

Case reports suggest an association between RDX exposure (via ingestion, inhalation, and possibly dermal exposure) and neurological effects (see Appendix C, Section C.2). Severe neurological disturbances include tonic-clonic seizures (formerly known as grand mal seizures) in factory workers (Testud et al., 1996a; Testud et al., 1996b; Kaplan et al., 1965; Barsotti and Crotti, 1949), seizures and convulsions in exposed soldiers serving in Vietnam (Ketel and Hughes, 1972; Knepshield and Stone, 1972; Hollander and Colbach, 1969; Stone et al., 1969; Merrill, 1968), seizures, dizziness, headache, and nausea following nonwartime/nonoccupational exposures (Kasuske et al., 2009; Davies et al., 2007; Küçükardali et al., 2003; Hett and Fichtner, 2002; Harrell-Bruder and Hutchins, 1995; Goldberg et al., 1992), and seizures in a child following ingestion of plasticized RDX from the mother's clothing (Woody et al., 1986).

Studies in Experimental Animals

Nervous system effects in experimental animals include a wide array of behavioral changes consistent with the induction of seizures by RDX exposure, and have been observed in the majority of chronic, subchronic, and developmental studies examining oral exposure to RDX (see Table 1-3

[5]The memory quotient index measured short-term hearing memory, visual memory, combined hearing and visual memory, and learning ability. The block design index measured visual perception and design replication, and the ability to analyze spatial relationships.

and Figure 1-1). Although study authors interchangeably used the terms seizures and convulsions, seizures, which result from abnormal electrical activity in the brain, can outwardly manifest in a variety of ways. While seizures can be detected in the form of convulsions, they can also manifest as facial twitches, tremors, or increased irritability, or they may go unnoticed. While behavioral methods exist to capture a spectrum of responses known to occur as a result of this aberrant neuronal activity, the most reliable detection methods are electrophysiological (Racine, 1972).

Convulsions have been reported in studies with different animal species and experimental designs. In every study that reported convulsions, the incidence of convulsions increased with dose. In 2-year dietary studies in rats (F344 and Sprague-Dawley) and mice ($B6C3F_1$), convulsions were observed beginning at doses of 35–40 mg/kg-day, but not at lower doses (Lish et al., 1984; Levine et al., 1983b; Hart, 1976).[6] Subchronic dietary exposure to RDX was also associated with convulsions in the rat, although doses reported to increase convulsive activity were inconsistent across studies. Convulsions were reported in RDX-exposed rats at subchronic doses as low as 8 and 25 mg/kg-day (Crouse et al., 2006; von Oettingen et al., 1949). In three other rat studies involving exposure durations of 30–90 days, no evidence of seizures, convulsions, or tremors was reported at doses ranging from 1 to 50 mg/kg-day (MacPhail et al., 1985; Cholakis et al., 1980) (both unpublished technical reports). Levine et al. (1990) reported convulsions in rats following subchronic exposure only at a dose of 600 mg/kg-day (a dose associated with 100% mortality); however, the unpublished technical report of this study (Levine et al., 1981a) inconsistently reported convulsions at 600 and ≥30 mg/kg-day, thereby reducing confidence in the identification of the dose level at which nervous system effects were observed in this study. RDX exposure (by gavage) during gestation in the rat was associated with induction of seizures or convulsions in the dams at doses ranging from 2 to 40 mg/kg-day (Angerhofer et al., 1986; Cholakis et al., 1980) (unpublished technical reports), demonstrating that effects on the nervous system can be observed following exposure durations as short as 10–14 days. Convulsions were also reported in dogs exposed to 50 mg/kg-day RDX for 6 weels (von Oettingen et al., 1949), but not 10 mg/kg-day for the 13 weeks (Hart, 1974) (unpublished technical report), and in five of six monkeys following a gavage dose of 10 mg/kg-day (Martin and Hart, 1974) (unpublished technical report).

In the only study addressing susceptibility to seizures, Burdette et al. (1988) found that seizure occurrence was more frequent in Long Evans rats exposed to a single dose of 50 or 60 mg/kg RDX by gavage when challenged with an audiogenic stimulus 8 and 16 hours after treatment. However, no audiogenic seizures were observed at the earlier 2- and 4-hour post-

[6]The 2-year dietary studies in F344 rats by Levine et al. (1983b) and $B6C3F_1$ mice by Lish et al. (1984) were available only as a laboratory reports. An external peer review was conducted by EPA in July 2012 to evaluate the accuracy of experimental procedures, results, and interpretation and discussion of the findings presented. A report of this peer review performed by Versar, Inc. is available through the EPA's IRIS Hotline at (202) 566-1676 (phone) or hotline.iris@epa.gov (e-mail address), and on the Health and Environmental Research Online (HERO) database (U.S. EPA, 2012d). The 2-year dietary study in Sprague-Dawley rats by Hart (1976) is available as an unpublished technical report.

dosing test periods even though RDX plasma concentrations were elevated throughout the testing period. In a complementary experiment, Long Evans rats treated daily with 6 mg/kg-day RDX for up to 18 days required fewer stimulation trials to exhibit amygdaloid kindled seizures compared to controls. Neither the purity nor the specific particle size of the RDX used in the experiments by Burdette et al. (1988) was reported.

The majority of animal studies reported convulsions and/or seizures as clinical observations; interpretation of these observations is limited to some extent because the nature and severity of convulsions and seizures were not more fully characterized. The 90-day study by Crouse et al. (2006)[7] was one of the few studies that collected and reported incidence data for convulsions and tremors, and demonstrated a clear dose-related increase in convulsions and tremors in male and female F344 rats associated with RDX exposure via gavage (see Table 1-3). Tremors were reported following administration of ≥12 mg/kg-day, persisting throughout the 90-day study. Convulsions were observed at ≥8 mg/kg-day in male and female rats; information on duration and onset was not reported (Crouse et al., 2006).

In general, gavage dosing induced convulsions at lower doses than did dietary administration. For example, in the gavage studies by Crouse et al. (2006) and Cholakis et al. (1980), convulsions were observed in 1–3 rats/group at doses of 2–8 mg/kg-day; at doses of 15–20 mg/kg-day, convulsions were observed in approximately 60–70% of the animals. In contrast, in a 2-year dietary study by Levine et al. (1983b), convulsions were reported only at a dose of 40 mg/kg-day; no convulsions were observed at lower doses (≤8 mg/kg-day). The difference in response between gavage and dietary administration may be due to the bolus dosing resulting from gavage administration and the comparatively faster absorption and higher peak blood concentrations of RDX.

Several experimental animal studies documented that unscheduled deaths were frequently preceded by convulsions or seizures. In a 2-year study in rats, Levine et al. (1983b) noted that tremors and/or convulsions were often seen in high-dose animals prior to their death. In a rat developmental toxicity study (Cholakis et al., 1980), investigators concluded that early deaths in dams were preceded by convulsions based on the observation of convulsions in one rat prior to death, and a similar appearance (e.g., dried blood around the mouth and nose) in other dams that died during the study. Convulsions preceding death were also observed in pregnant Sprague-Dawley rats exposed to RDX during gestation (Angerhofer et al., 1986).

[7]The 13-week gavage study in F344 rats by Crouse et al. (2006) was available only as a laboratory report. An external peer review was conducted by Versar, Inc. in July 2012 to evaluate the accuracy of experimental procedures, results, and interpretation and discussion of the findings presented. The U.S. EPA (2012d) report of this peer review is available through the EPA's IRIS Hotline at (202) 566-1676 (phone) or hotline.iris@epa.gov (e-mail address), and on the HERO database.

The 90-day Crouse et al. (2006) study provides the most detailed information on the relationship between convulsions and mortality (see Appendix C, Table C-10 for additional information on evidence of mortality associated with RDX exposure). Convulsions (3/20) and pre-term deaths (2/20)[8] were observed in male and female rats exposed to 8 mg/kg-day RDX; the incidences of both convulsions and pre-term deaths were higher in dose groups with greater exposures. Investigators stated that nearly all observed pre-term deaths in rats exposed to the three higher doses (10, 12, and 15 mg/kg-day RDX) for 90 days were preceded by neurotoxic signs such as rearing behavior, tremors and convulsions; however, pre-term death did not occur in all animals that convulsed. Some uncertainty exists in that convulsions were not typically observed during a functional observational battery (FOB) test conducted after exposure, likely due to the time needed to complete exposures prior to beginning behavioral testing. Of the 100 RDX-treated rats in the Crouse et al. (2006) study, convulsions were documented in 34 male and female rats across the five dose groups (with convulsions initially observed anywhere from day 7 to 87); based on additional information provided as a memorandum by study investigators (Johnson, 2015a), 26 of these 34 rats (76%) survived to the end of the 90-day study. In general, higher doses of RDX were associated with fewer days of exposure before the first convulsion was observed. Of the eight rats that exhibited convulsions prior to pre-term death, convulsions were documented anywhere from the same day that the animal died to 8 weeks prior to death. Of the 26 rats that seized and survived to day 90, the first seizures were observed as early as day 10 and as late as day 87. Thus, while an increase in mortality was observed in the Crouse et al. (2006) study at the same dose as convulsions, the additional information provided by Johnson (2015a) do not show as clear a correspondence between convulsions (and other neurotoxic signs) and mortality. Analysis of these data is limited to the extent that convulsions may have occurred at times when animals were not observed and are therefore undercounted in the individual animal data; however, Johnson (2015a) noted that it is unlikely that seizure observations were missed, since seizures generally occurred soon after dosing.

A few studies reported mortality that was not specifically or directly associated with neurological effects (see Appendix C, Table C-10) (Angerhofer et al., 1986; Levine et al., 1981a; von Oettingen et al., 1949); however, in these studies, animals may not have been monitored for clinical observations with sufficient frequency to have observed convulsive activity prior to death. In case reports of convulsions and other nervous system effects in workers exposed to RDX during manufacture and in individuals exposed acutely as a result of accidental or intentional ingestion, there were no reports of mortality subsequent to the convulsions (see Appendix C, Section C.2).

Additional neurobehavioral effects associated with RDX exposure in rats included increased hyperactivity, hyper-reactivity to approach, fighting, and irritability at doses similar to those that

[8]At the 8 mg/kg-day dose level, the three rats that convulsed survived to the end of the study; no convulsions were observed in the two rats that died before study termination.

induced tremors, convulsions, and seizures (20–100 mg/kg-day) (Levine et al., 1990; Angerhofer et al., 1986; Levine et al., 1983b; Levine et al., 1981a, b; von Oettingen et al., 1949). Hyperactivity and nervousness were also reported in male mice that received a subchronic exposure to 320 mg/kg-day RDX (Cholakis et al., 1980). No changes in motor activity, flavor aversion, scheduled-controlled behavior, or acoustic startle response were observed in a 30-day gavage study in rats (changes in acoustic startle response in acute exposures at higher doses [12.5–50 mg/kg] were noted), but doses were relatively low (≤10 mg/kg-day) (MacPhail et al., 1985), and no significant changes in behavioral or neuromuscular activity were observed in rats following exposure to ≤15 mg/kg-day for 90 days (Crouse et al., 2006). Crouse et al. (2006) observed that stained haircoats and increased barbering in female F344 rats receiving 15 mg/kg-day may have been caused by the oral dosing procedure (gavage) alone.

Changes in absolute and relative brain weight were mixed across studies. Elevated absolute brain weights were reported in subchronic assays in B6C3F$_1$ mice and F344 rats (Crouse et al., 2006; Levine et al., 1990; Levine et al., 1981a, b; Cholakis et al., 1980); however, the changes were not consistently observed across studies. Relative brain weights in some studies showed correspondingly greater increases compared to absolute brain weight (Crouse et al., 2006; Levine et al., 1983b; Cholakis et al., 1980), but these changes were likely a result of changes in body weight in the study, and were not a useful measure of effects of RDX on brain weights. In 2-year oral studies, a decrease in absolute brain weight of female B6C3F$_1$ mice (3–4% relative to control) was reported at doses ≥35 mg/kg-day (Lish et al., 1984), where as an increase in absolute brain weight (2% relative to control) was observed in F344 rats at a dose of 40 mg/kg-day (Levine et al., 1983b). Less weight is placed on evidence of organ weight changes from chronic (2-year) studies because normal physiological changes associated with aging and intercurrent disease may contribute to inter-animal variability that could confound organ weight interpretation (Sellers et al., 2007).

In some studies, seizures appeared soon after dosing, suggesting that seizure induction was more strongly correlated with dose level than with duration of exposure. Consistent with this observation are the findings of Williams et al. (2011), who demonstrated that RDX is rapidly absorbed and crosses the blood:brain barrier following oral administration in rats, and that distribution of RDX (8 µg/g wet weight) to the brain correlated with seizure onset.

While a dose-response relationship was observed consistently within studies, a dose that induced convulsions in animals in one study did not necessarily induce convulsions at the same dose in another study. This lack of consistency may be attributed, at least in part, to differences in the purity or particle size of the test material across studies. Assuming that increased particle size (and the corresponding reduction in available surface area compared with smaller particle sizes) results in slowed absorption and distribution to the brain, studies that used a larger particle size may be expected to produce less neurotoxicity in test animals. The mouse study by Cholakis et al. (1980) used a relatively large RDX particle size (200 µm) compared to the rat study by Levine et al. (1983b) that used a smaller (<66 µm) particle size. This could contribute to why the Cholakis et al.

(1980) subchronic dietary study in the mouse (doses up to 320 mg/kg-day RDX) and rat (doses up to 40 mg/kg-day) failed to report seizures or convulsions. Finally, differences in study design may have contributed to differences in reported neurological responses in subchronic and chronic duration studies. In particular, the protocols for observation for clinical signs (e.g., observations performed once daily in the morning in Levine et al. (1983b)) may not have been sufficiently frequent to accurately measure the incidence of seizures or other nervous system effects.

Table 1-2. Evidence pertaining to nervous system effects in humans

Reference and study design	Results			
Ma and Li (1993) (China) Cross-sectional study, 60 workers from the same plant exposed to RDX (30 in Group A [26 males; 4 females]; 30 in Group B [24 males; 6 females]), compared to 32 workers with similar age, education level, and length of employment from same plant with no exposure to RDX (27 males; 5 females). **Exposure measures**: Details of exposure measurement were not provided; two groups of workers exposed to the following mean RDX concentrations in air (basis for dividing workers into two exposure groups was not provided). **Concentration** (mg/m^3) (mean ± standard deviation): Group A 0.407 (± 0.332) Group B 0.672 (± 0.556) **Effect measures**[a]: Five neurobehavioral function tests and five additional memory subtests. **Analysis**: Variance (F-test); unadjusted linear regression, multiple regression, and correlation analysis.	**Neurobehavioral function tests, scaled scores** *(mean, standard deviation)*			
	Test	Control	Group A	Group B
	Memory retention*	111.3 (9.3)	96.9 (9.6)	91.1 (10.3)
	Simple reaction time (milliseconds)	493 (199)	539 (183)	578 (280)
	Choice reaction time (milliseconds)	763 (180)	775 (161)	770 (193)
	Block design* (elapsed time)	18.0 (5.4)	16.0 (4.3)	13.5(6.7)
	Letter cancellation (quality per unit time)	1,487 (343)	1,449 (331)	1,484 (443)
	*$p < 0.01$ (overall F-test); no statistically significant differences between Group A and Group B. Lower score indicates worse performance.			
	Memory retention subtests, scaled scores *(mean, standard deviation)*			
	Subtest	Control	Group A	Group B
	Directional memory*	23.5 (3.6)	17.2 (4.9)	18.1 (5.7)
	Associative learning*	24.9 (5.1)	20.0 (4.3)	18.5 (4.6)
	Image free recall*	24.1 (3.8)	20.9 (4.1)	20.4 (3.3)
	Recognition of nonsense pictures*	26.3 (3.6)	23.2 (4.9)	21.6 (4.3)
	Associative recall of portrait characteristics*	26.3 (3.3)	20.3 (4.4)	18.5 (4.3)
	*$p < 0.01$ (overall F-test); no statistically significant differences between Group A and Group B. Lower score indicates worse performance. Total behavioral score negatively correlated with exposure index (high exposure correlated with poor performance).			

[a]Symptom data were not included in evidence table because of incomplete reporting.

Table 1-3. Evidence pertaining to nervous system effects in animals

Reference and study design	Results
Convulsions and neurobehavioral effects	
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	One male in the 35 mg/kg-d dose group and one female in the 175/100 mg/kg-d group convulsed near the end of the study.
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	No nervous system effects, as evidenced by clinical signs or changes in appearance or behavior, were reported.
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Tremors, convulsions, and hyper-responsiveness to stimuli were noted in males and females at 40 mg/kg-d; no incidence data were reported.
Cholakis et al. (1980) Mice, B6C3F$_1$, 10–12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 40, 60, or 80 mg/kg-d for 2 wks followed by 0, 320, 160, or 80 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Hyperactivity and/or nervousness observed in 50% of the high-dose males; no signs observed in females[b]; no incidence data were reported.
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	No nervous system effects, as evidenced by clinical signs or changes in appearance or behavior, were reported.

Reference and study design	Results						
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26/sex/group; F2: 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	No nervous system effects were reported.						
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8[b]	10	12	15
	Convulsions *(incidence)*						
	M	0/10	0/10	1/10	3/10	8/10	7/10
	F	0/10	0/10	2/10	3/10	5/10	5/10
	Tremors *(incidence)*						
	M	0/10	0/10	0/10	0/10	2/10	3/10
	F	0/10	0/10	0/10	0/10	0/10	1/10
(Levine et al. (1990); Levine et al. (1981a), 1981b))[c] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Hyper-reactivity to approach was observed in rats (sex not specified) receiving ≥100 mg/kg-d; no incidence data were reported. Tremors and convulsions were observed prior to death in one female and two male rats receiving 600 mg/kg-d.[d] (600 mg/kg-d was lethal to all rats.)						
von Oettingen et al. (1949) Rats, sex/strain not specified, 20/group 90–97% pure, with 3–10% HMX; particle size not specified 0, 15, 25, or 50 mg/kg-d Diet 13 wks	Hyperirritability and convulsions were observed in the 25 and 50 mg/kg-d groups[b]; no incidence data were reported.						

Reference and study design	Results				
Hart (1974) Dogs, Beagle, 3/sex/group Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	No nervous system effects, as evidenced by clinical signs or changes in appearance or behavior, were reported.				
Martin and Hart (1974) Monkeys, Cynomolgus or Rhesus [e], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	Doses	0	0.1	1	10[b]
	CNS effects characterized as depression, trembling, shaking, jerking, or convulsions *(incidence)*				
	M	0/3	0/3	0/3	2/3
	F	0/3	0/3	0/3	3/3
von Oettingen et al. (1949) Dogs, breed not specified, 5 females/group (control); 7 females/group (exposed) 90–97% pure, with 3–10% HMX; particle size not specified 0 or 50 mg/kg-d Diet 6 d/wk for 6 wks	Treated dogs exhibited convulsions, excitability, ataxia, and hyperactive reflexes[b]; no incidence data were reported.				
MacPhail et al. (1985) Rats, Sprague-Dawley derived CD, 8–10 males or females/group Purity 84 ± 4.7%; ≤66 µm particle size 0, 1, 3, or 10 mg/kg-d Gavage 30 d	No changes in motor activity, flavor aversion, scheduled-controlled response, or acoustic startle-response were reported.				
Cholakis et al. (1980) Rats, F344, 24–25 females/group 88.6% pure, with 9% HMX and 2.2% water as contaminants 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 6–19	Doses	0	0.2	2.0	20
	Convulsions *(incidence)*				
	F	0/24	0/24	1/24	18/25
Angerhofer et al. (1986) (range-finding study) Rats, Sprague-Dawley, 6 pregnant females/group Purity 90%; 10% HMX and 0.3% acetic acid occurred as contaminants 0, 10, 20, 40, 80, or 120 mg/kg-d Gavage GDs 6–15	Convulsions preceding death were observed at ≥40 mg/kg-d; no incidence data were reported.				

Reference and study design	Results						
Angerhofer et al. (1986) Rats, Sprague-Dawley, 39–51 mated females/group Purity 90%; 10% HMX and 0.3% acetic acid occurred as contaminants 0, 2, 6, or 20 mg/kg-d Gavage GDs 6–15	Convulsions and hyperactivity[b] were observed at 20 mg/kg-d; no incidence data were reported.						
Burdette et al. (1988) Rats, Long Evans, 10–21 males/group 0, 10, 12.5, 20, 25, 50, or 60 mg/kg-d Study conducted as two experiments with the same study design, each with a control group Gavage 8 hrs (single exposure) After an 8-hr exposure, rats placed in observation chamber; 0–64 kHz, 95 dB ultrasonic cleaner turned on for 1 min or until seizure initiated with uncontrolled running (whichever occurred first)	Doses	0	10	12.5	20	25	50	60
	Prevalence of audiogenic seizures *(%)*[†]							
	M	0	9	0	29	40	82*	78*
	[†]Values estimated from graph using Grab It! Software. Statistical significance indicated by study authors; $p < 0.017$.							
Brain weight								
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7	35	175/100		
	Absolute brain weight *(percent change compared to control)*							
	M	0%	−0.2%	0.61%	0.81%	−1%		
	F	0%	−2%	−2%	−4%*	−3%*		
	Relative brain weight *(percent change compared to control)*							
	M	0%	4%	2%	2%	5%		
	F	0%	−4%	−1%	−3%	18%*		
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8	40		
	Absolute brain weight *(percent change compared to control)*							
	M	0%	2%	−1%	2%	2%		
	F	0%	−0.3%	−0.4%	1%	2%*		
	Relative brain weight *(percent change compared to control)*							
	M	0%	0%	8%	2%	22%*		
	F	0%	−1%	3%	4%	20%*		

Reference and study design	Results						
Cholakis et al. (1980) Mice, B6C3F$_1$, 10–12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size Experiment 1: 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute brain weight *(percent change compared to control)*						
	M	0%	–	–	–	2%	2%
	F	0%	–	–	–	4%	2%
	Relative brain weight *(percent change compared to control)*						
	M	0%	–	–	–	6%	2%
	F	0%	–	–	–	0%	3%
Experiment 2: 0, 40, 60, or 80 mg/kg-d for 2 wks followed by 0, 320, 160, or 80 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Doses	0		80	160		320
	Absolute brain weight *(percent change compared to control)*						
	M	0%		0%	2%		10%
	F	0%		0%	4%		2%
	Relative brain weight *(percent change compared to control)*						
	M	0%		−3%	1%		8%
	F	0%		0%	3%		−4%
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute brain weight *(percent change compared to control)*						
	M	0%	–	–	–	3%	0%
	F	0%	–	–	–	0%	0%
	Relative brain weight *(percent change compared to control)*						
	M	0%	–	–	–	7%*	10%*
	F	0%	–	–	–	5%	6%
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Absolute brain weight *(percent change compared to control)*						
	M	0%	−1%	−0.3%	2%	5%*	7%*
	F	0%	−2%	6%	1%	4%	6%
	Relative brain weight *(percent change compared to control)*						
	M	0%	6%	10%	5%	3%	4%
	F	0%	−2%	−2%	−12%*	−12%*	−15%*

Reference and study design	Results						
(Levine et al. (1990); Levine et al. (1981a), 1981b))[c] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Doses	0	10	30	100	300	600
	Absolute brain weight *(percent change compared to control)*						
	M	0%	1%	0.53%	−6%	–	–
	F	0%	−1%	1%	2%	–	–
	Relative brain weight *(percent change compared to control)*						
	M	0%	4%	7%	14%	–	–
	F	0%	0.3%	2%	5%	–	–

*Statistically significant ($p < 0.05$) based on analysis by study authors.
[a]Doses were calculated by the study authors.
[b]Mortality was reported in some RDX-treated groups in this study.
[c]Levine et al. (1981a) is a laboratory report of a 13-week study of RDX in F344 rats; two subsequently published papers (Levine et al., 1990; Levine et al., 1981b) present subsets of the data provided in the full laboratory report.
[d]Discrepancies in the doses at which convulsions occurred were identified in the technical report. The nervous system effects reported in this table and in the corresponding exposure-response array are those provided in the results section of the technical report (Levine et al., 1981a) and in the published paper (Levine et al., 1990). In other sections of the technical report, the authors reported that hyperactivity to approach and convulsions were observed in rats receiving ≥30 mg/kg-day (abstract and executive summary), or that mortality was observed in rats receiving 100 mg/kg-day and that hyperactivity to approach, tremors, and convulsions were observed in animals exposed to lethal doses (discussion).
[e]The species of monkey used in this study was inconsistently reported in the study as either Cynomolgus (in the methods section) or Rhesus (in the summary).

CNS = central nervous system; GD = gestational day; HMX = octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine; TWA = time-weighted average

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.

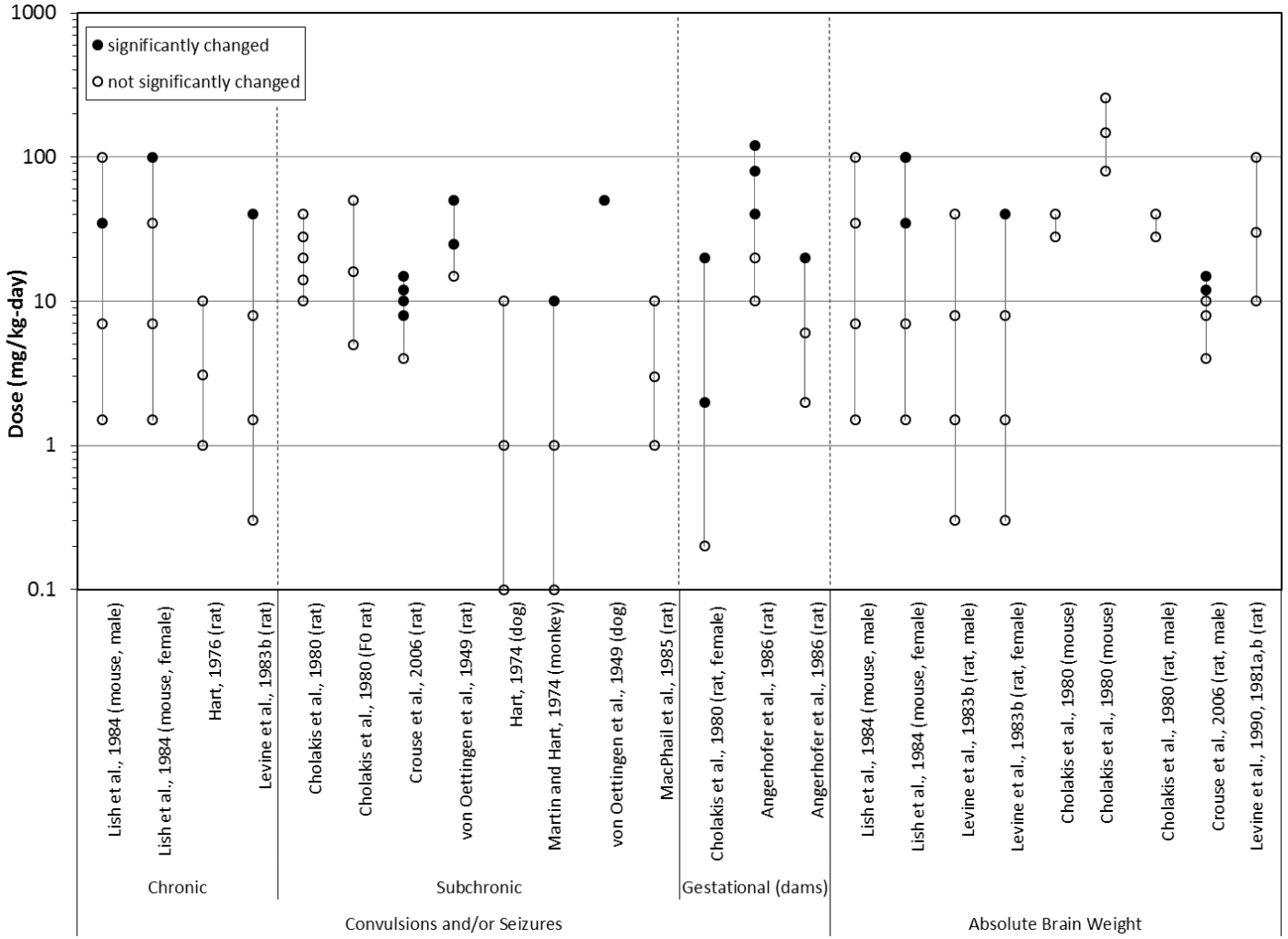


Figure 1-1. Exposure response array of nervous system effects following oral exposure.[1]

[1]Because convulsions and seizures are rare in experimental animals, any occurrence in an RDX-exposed group was considered treatment-related. Given the severity of this endpoint, a response in treated groups was determined to be significant (filled circles) in the array where there was any occurrence of convulsions and/or seizures reported in the study, whether or not the incidence was statistically significantly elevated over the control.

Mechanistic Evidence

Studies that have explored the mode of action (MOA) of RDX on the central nervous system (CNS) have focused on the potential impacts on neurotransmission. These studies implicate a MOA for RDX-induced seizures and convulsions involving distribution to the brain (across the blood:brain barrier) and subsequent effects on neurotransmitters, including gamma-amino butyric acid (GABA) and glutamate. There is significant evidence from the scientific literature to suggest that RDX neurotoxicity results from interactions of RDX with the $GABA_A$ receptor. GABA is a major inhibitory neurotransmitter in the brain, and the $GABA_A$ receptor has been implicated in susceptibility to seizures (Galanopoulou, 2008).

In research conducted by the U.S. Army Center for Health Promotion and Preventative Medicine, Williams et al. (2011) and Bannon et al. (2009a) showed a correlation between blood and brain concentrations of RDX in rats that received a single oral dose of RDX (>98–99.5% purity) by gavage, which closely correlated with the time of seizure onset. RDX (75 mg/kg) was distributed to the brain in direct proportion to levels found in the blood, while time to seizure onset was reduced as RDX brain levels increased (Williams et al., 2011). Similarly, oral exposure to RDX (via a gel capsule: 3 or 18 mg/kg) resulted in quick absorption followed by transport to the brain and subsequent alterations in neurotransmission (Bannon et al., 2009a).

In receptor binding studies, RDX has only showed affinity for $GABA_A$ receptors (Williams et al., 2011; Williams and Bannon, 2009). Specifically, RDX showed a significant affinity for the picrotoxin convulsant site of the GABA channel. RDX treatment in brain slices from the basolateral amygdala inhibited $GABA_A$-mediated inhibitory postsynaptic currents and initiated seizure-like neuronal discharges. RDX exposure may reduce the inhibitory effects of GABAergic neurons, resulting in enhanced excitability that could lead to seizures (Williams et al., 2011; Williams and Bannon, 2009). The limbic system, and the amygdala and hippocampus in particular, are known to be critical to the development of seizures in various human conditions (e.g., epilepsy) and animal models (Jefferys et al., 2012; Gilbert, 1994). Burdette et al. (1988) hypothesized that the limbic system was involved in seizures caused by RDX exposure, given that amygdaloid kindled rats (rats subjected to patterns of electrical stimulation to promote the development of seizures) exhibited pro-convulsant activity at a dose that was approximately half of the dose necessary for RDX to induce spontaneous seizures (rats treated with RDX also required fewer electrical stimulations to trigger kindled seizures). Potential limbic system involvement is also suggested given its role in integrating emotional and behavioral responses (including aggression) and the anecdotal observations of hyperactivity, hyper-responsiveness to approach, and irritability noted across several studies of RDX toxicity (Levine et al., 1990; Levine et al., 1983b; Levine et al., 1981a, b; Cholakis et al., 1980; von Oettingen et al., 1949).

RDX binding at the picrotoxin convulsant site of the GABA channel may also inform the relationship between exposure to the chemical and the time when a seizure was observed. In general across the RDX database, induction of convulsions and seizures appears to be more strongly

correlated with dose than duration of exposure. However, Gerkin et al. (2010) demonstrated that young C57/Bl6 mice injected intraperitoneally (i.p.) with picrotoxin to induce seizures had a significantly increased frequency of elevated neuronal activity ("Up state"), and firing rates were significantly increased in neocortical neurons up to 24 hours after exposure. It is possible that this period of elevated neuronal activity could increase the likelihood that a subsequent stimulus could trigger a seizure. While the study authors did not look at longer durations post exposure, it is possible in a chronic exposure scenario that repeated exposure to RDX binding at the same site as picrotoxin, through a general increase in brain tissue with elevated neuronal activity, could increase the likelihood of seizures developing over time, or have other longer-term effects on normal brain function.

It is possible to construct a hypothetical MOA for RDX-induced neurotoxicity based on the evidence summarized above. Following absorption and transport to the brain:

1) Parent RDX acts as a receptor antagonist (supported by Schneider et al. (1977) and Williams et al. (2011)), binding noncompetitively to the picrotoxin convulsant site of the $GABA_A$ receptor (supported by Williams and Bannon (2009) and Williams et al. (2011)).
2) RDX binding to the $GABA_A$ receptor results in decreased conduction of chloride through the ion channel.
3) Reduced chloride conduction results in depolarization of the neuronal membrane, thereby reducing spontaneous inhibitory postsynaptic currents (sIPSCs). Williams et al. (2011) observed a reduction in the amplitude and frequency of sIPSCs in whole-cell in vitro recordings of neurons in brain slices from the rat basolateral amygdala after exposure to RDX. In addition, RDX treatment of slices inhibited GABA-induced currents.
4) Reduction in sIPSCs results in an overall reduction in inhibitory inputs to the nervous system. Williams et al. (2011) observed a pattern of seizure-like neuronal discharges in vitro from slices of the basolateral amygdala in rats after adding RDX and noted that the effects were not reversible after 40 minutes of washout.

The steps above provide a biologically plausible sequence of mechanistic events that result in the generation of seizure-like neuronal activity. Reduction of the inhibitory GABAergic signaling is common to many convulsants, as summarized in Kalueff (2007). Some organochlorine insecticides, including alpha-endosulfan, dieldrin, and lindane, also exert neurotoxic effects through interaction with the $GABA_A$ receptor, and can produce a range of hyperexcitability effects (including convulsions) in mammals (Vale et al., 2003; Bloomquist, 1992; Suñol et al., 1989). The interaction of RDX with the $GABA_A$ receptor is directly supported by receptor-binding assays (Williams et al., 2011). Although these binding assays were performed on rat receptors, it is plausible that the results are relevant to human neurotoxicity. Seizures have been observed in many species, including humans, rats, mice, dogs, lizards, and birds at varying dosages and durations of exposure

(Quinn et al., 2013; McFarland et al., 2009; Johnson et al., 2007; Bruchim et al., 2005; Küçükardali et al., 2003; Woody et al., 1986; Lish et al., 1984; Berry et al., 1983; Levine et al., 1983b). A more recent meta-analysis of toxicogenomic data across a phylogenetically diverse set of organisms (rat, quail, fathead minnow, earthworm, and coral) demonstrated that neurotoxic responses are conserved in more highly-related species and that binding to the $GABA_A$ receptor is a common molecular initiating event (Garcia-Reyero et al., 2011). While these lines of evidence do not preclude a role of other receptors as yet unscreened for RDX binding affinity, they support involvement of the GABAergic pathway described above in the development of RDX neurotoxicity.

The $GABA_A$ receptor is also a target of many anticonvulsant therapies (e.g., benzodiazepines, propofol, barbiturates) (Meldrum and Rogawski, 2007; Möhler, 2006). Additional support for the involvement of GABAergic signaling in the neurotoxicity of RDX comes from human case reports. In multiple case reports, medical intervention included treatment with benzodiazepines (commonly diazepam or lorazepam) to treat seizing patients (Kasuske et al., 2009; Davies et al., 2007; Küçükardali et al., 2003; Hett and Fichtner, 2002; Woody et al., 1986). Benzodiazepines act in large part by enhancing the effects of GABA at the $GABA_A$ receptor by increasing chloride conductance, resulting in anticonvulsant and relaxant effects (Goodman et al., 1996).

Some other pro-convulsant agents with minimal direct toxicity to nerve cells, such as sarin and some organophosphate pesticides, are known to act through inhibition of acetylcholinesterase (AChE) activity (McDonough and Shih, 1997). Some of the clinical signs observed following RDX exposure are similar to the clinical signs associated with organophosphate pesticides and nerve agents (Crouse et al., 2006; Burdette et al., 1988; Barsotti and Crotti, 1949). However, the limited data available for RDX do not support AChE inhibition as a primary mechanism because: (1) blood and brain levels of AChE are unaffected by RDX (Williams et al., 2011; Williams and Bannon, 2009); and (2) in vitro neurotransmitter receptor binding studies do not reveal any affinity of RDX for acetylcholine receptors (Williams et al., 2011; Williams and Bannon, 2009). Additionally, common AChE-induced symptoms (salivation and lacrimation) have not routinely been observed (Williams et al., 2011). RDX showed no affinity for other receptors that are known targets of convulsants, including the glutamate family of receptors, nicotinic receptors, glycine receptors, and several monoamine receptors (Williams et al., 2011; Williams and Bannon, 2009).

In a microarray experiment, Bannon et al. (2009a) found that RDX caused a down regulation of an abundance of genes in the cerebral cortex related to neurotransmission, including those encoding proteins involved in synaptic transmission and vesicle transport. Genes encoding proteins involved in the glutamate pathway were also underexpressed, indicating a possible mechanism of action for RDX via excessive glutamate stimulation. The authors speculated that this depression of the major excitatory neurotransmitter system could be a negative response to the increase in seizure likelihood from RDX influx into the brain. Molecular changes in response to RDX have been described by Zhang and Pan (2009b), who observed significant changes in micro-RNA (miRNA) expression in the brains of $B6C3F_1$ mice fed 5 mg RDX/kg diet (estimated dose: 0.75–1.5

mg/kg-day; Bannon et al., 2009b) for 28 days. One miRNA, miR-206, was upregulated 26-fold in RDX-exposed brains; brain-derived neurotrophic factor (BDNF) was identified as a downstream gene target of this miRNA, along with two other miRNAs that were upregulated in RDX-exposed brains (miR-30a and miR-195) (Zhang and Pan, 2009a, b). BDNF is a member of the neurotrophin family of growth factors, and promotes the survival and differentiation of existing and new neurons. Deng et al. (2014) conducted miRNA and mRNA profiling in rats to identify targets up or downregulated after 48-hour exposure to RDX, finding that many of the gene targets of these miRNAs were associated with nervous system function, and may contribute to the neurotoxicity of RDX. However, while effects of RDX on BDNF expression or other downstream targets may play a role in RDX neurotoxicity, the utility of miRNAs as predictors of toxicity has not been demonstrated and downstream targets of miRNA require verification (Bannon et al., 2009b). Thus, the contribution, if any, of aberrant expression of a suite of miRNAs to the MOA for RDX neurotoxicity is unknown.

Recent research has provided greater insight to inform a mechanistic basis of RDX neurotoxicity. While other possible MOA(s) may contribute to the overall neurotoxicity of RDX, the demonstrated affinity of RDX for the $GABA_A$ receptor, evidence of supportive electrophysiological changes with direct application of RDX, and toxicokinetic evidence of distribution of RDX to the brain provides a mechanistic basis for the association of seizures with exposure to RDX. The available information supports that RDX-induced hyperactivity and seizures likely result from inhibition of GABAergic signaling in the limbic system.

Integration of Nervous System Effects

Evidence for nervous system effects associated with exposure to RDX comes from studies in both humans and animals. One occupational study reported memory impairment and decrements in certain neurobehavioral tests in workers exposed to RDX compared to controls (Ma and Li, 1993), and human case reports provide other evidence of an association between acute RDX exposure and neurological effects. There was consistent evidence of neurotoxicity associated with exposure to RDX; 11 of 16 repeat-dose animal studies (of varying design) reported neurological effects (some severe), including seizures, convulsions, tremors, hyperirritability, hyper-reactivity, and behavioral changes, associated with RDX exposure (Crouse et al., 2006; Angerhofer et al., 1986; Levine et al., 1983b; Levine et al., 1981b; Cholakis et al., 1980; von Oettingen et al., 1949). In most of these studies, the occurrence of neurological effects was dose-related. In those studies that found no evidence of RDX-associated neurotoxicity (MacPhail et al., 1985; Cholakis et al., 1980; Hart, 1976, 1974), differences in dosing, particle size, and purity of the RDX administered could possibly account for the lack of effect. Seizures resulting from RDX exposure likely result from inhibition of GABAergic signaling due to the interaction of RDX with the $GABA_A$ receptor. Convulsant receptor binding leading to a decreased seizure threshold, considered with kindling studies, suggests that the effect is specific to CNS toxicity.

Together, toxicological information in animals and humans, supported by toxicokinetic and mechanistic information, provides a coherent identification of nervous system effects as a human hazard of RDX exposure.

1.2.2. Kidney and Other Urogenital System Effects

The association between RDX exposure and effects on clinical measures of kidney function was examined in one occupational epidemiology study. Case reports, involving accidental exposure to ingested or inhaled RDX, offer some information on the potential for acute exposure to RDX to affect the kidney in humans. Organ weight and histopathology findings from experimental animal studies involving subchronic and chronic exposure to ingested RDX also provide data relevant to an examination of the association between RDX exposure and kidney and other urogenital system effects. A summary of these effects associated with RDX exposure is presented in Tables 1-4 to 1-8 and Figure 1-2. Experimental animal studies are ordered in the evidence table and exposure-response array by duration of exposure and then by species.

Human case reports of individuals accidently exposed to unknown amounts of RDX by ingestion or inhalation provide some evidence that RDX may affect the kidney and the urogenital system. Reported symptoms included decreased urine output (Ketel and Hughes, 1972; Knepshield and Stone, 1972; Hollander and Colbach, 1969; Merrill, 1968), blood in urine (Kasuske et al., 2009; Knepshield and Stone, 1972; Hollander and Colbach, 1969; Merrill, 1968), proteinuria (Kasuske et al., 2009; Küçükardali et al., 2003; Ketel and Hughes, 1972; Hollander and Colbach, 1969; Merrill, 1968), glucosuria (Küçükardali et al., 2003), elevated blood urea nitrogen (BUN) levels (Hollander and Colbach, 1969; Merrill, 1968), and one case of acute renal failure requiring hemodialysis following accidental inhalation of RDX (Ketel and Hughes, 1972). In many of these case reports, renal parameters returned to normal within a few days following exposure. No changes in renal parameters were reported in other individuals exposed to unknown amounts of RDX (Stone et al., 1969; Kaplan et al., 1965). In a cross-sectional epidemiologic study of workers from five U.S. Army munitions plants (69 exposed to RDX alone and 24 exposed to RDX and octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX); RDX exposure range: undetectable [<0.01 mg/m^3] to 1.6 mg/m^3), no statistically significant differences in BUN or total serum protein between nonexposed and RDX-exposed groups were observed (Hathaway and Buck, 1977) (Table 1-4). As it is a cross-sectional study, no information was provided on the length of employment or other proxies that could be used to indicate cumulative exposure concentrations.

Studies in experimental animals provide some evidence that RDX exposure is associated with kidney and other urogenital system effects (Table 1-5 and Figure 1-2). Histopathological changes in the urogenital system were associated with exposure to RDX in a 2-year bioassay. Specifically, increased incidences of kidney medullary papillary necrosis and pyelitis, uremic mineralization, bladder distention and/or cystitis, and suppurative prostatitis were observed in high-dose (40 mg/kg-day) male rats that died spontaneously or were sacrificed in moribund condition (Levine et al., 1983b). These renal effects were considered the principal cause of

treatment-related morbidity and mortality in these high-dose males. Similar kidney lesions were not observed in female rats in this study. An increased incidence of tubular nephrosis was observed in male B6C3F$_1$ mice exposed to 320 mg/kg-day RDX in feed for 90 days, but not in female mice in this study (Cholakis et al., 1980). In other chronic and subchronic oral studies in rats and mice, no histopathological changes in the kidney were associated with RDX exposure (Crouse et al., 2006; Levine et al., 1990; Lish et al., 1984; Levine et al., 1981a, b; Cholakis et al., 1980; Hart, 1976). Increased incidence of minimal to mild mineralization of the medulla was observed in male and female monkeys exposed to 10 mg/kg-day RDX for 90 days by gavage (Martin and Hart, 1974), but the study authors did not identify this as treatment related. No dose-related histopathological changes were reported in a subchronic study in dogs (Hart, 1974), and no histological alterations were noted in the kidneys of rabbits exposed dermally to a cumulative dose of 165 mg/kg RDX in dimethylsulfoxide (DMSO) received over a 4-week period (5 days/week) (McNamara et al., 1974). Measurement of serum chemistry parameters that may indicate effects on renal function, including BUN and uric acid, in studies of RDX in mice, rats, dogs, and monkeys (Crouse et al., 2008; Levine et al., 1990; Lish et al., 1984; Levine et al., 1981a, b; Cholakis et al., 1980; Hart, 1976, 1974; Martin and Hart, 1974) revealed variations (increases or decreases) from the respective control groups that were not dose-related.

The findings of suppurative prostatitis provide the strongest evidence of urogenital toxicity. A significant, dose-related increase in the total incidence of suppurative prostatitis was reported in male F344 rats exposed to ≥1.5 mg/kg-day RDX in the diet for 2 years (Levine et al., 1983b). Suppurative prostatitis was not observed in 90-day studies in the rat involving oral (dietary or gavage) exposure to RDX (Crouse et al., 2006; Levine et al., 1990; Levine et al., 1981a, b). Similarly, prostate effects were not observed in a 2-year dietary study in mice (Lish et al., 1984). The Levine et al. (1983b) report is the only 2-year study that reported examination of the prostate in rats. Some reports have hypothesized that the observation of prostate inflammation in Levine et al. (1983b) is secondary to a bacterial infection unrelated to RDX toxicity (ATSDR, 2012; Sweeney et al., 2012a; Crouse et al., 2006). For example, in describing the results from the 2-year dietary study in rats, Crouse et al. (2006) observed that the inflammation reflects a common condition in rodents, noting that since 85% of the incidence occurred in rats found at spontaneous death or moribund sacrifice (SDMS), it was most likely that the condition was a result of an incidental bacterial infection. However, Levine et al. (1983b) distinguished between nonsuppurative and suppurative inflammation (the latter being characterized by the formation of pus and a high concentration of neutrophils). Although the proportion of suppurative prostatitis was higher in SDMS rats, there was an increasing trend with dose in both the scheduled sacrifice (SS) and SDMS groups; the incidence of suppurative prostatitis in the control group was 4% when the SS and SDMS groups were combined. Additionally, the dose-related nature of the increased incidence suggests that the primary cause (potentially leading to bacterial infection) was treatment-related, as a more uniform distribution of rats with suppurative prostatitis would be expected with a spontaneous or age-

related lesion. The dose-responsiveness could be explained if the infections were secondary to treatment-related immunotoxicity, but there is no evidence from Levine et al. (1983b) to support this possibility. A more thorough analysis of immune endpoints in a 90-day gavage exposure of F344 rats did not identify any immunotoxic effects associated with RDX (Crouse et al., 2006).

As noted above, Levine et al. (1983b) documented an array of kidney and other urogenital lesions in their 2-year dietary exposure of F344 rats to RDX. However, the sequence by which those effects may have occurred is unclear. Renal medullary necrosis, bladder distension, and cystitis were observed mainly in the male rats exposed to 40 mg/kg-day RDX for 24 months, although one rat in the 0.3 mg/kg-day dose group also exhibited these lesions. Treatment-related effects on the kidney (necrosis) and bladder (distension/obstruction and hemorrhagic cystitis) were also identified in the 12-month pathology report (see Tables 1-6 to 1-8). The absence of these observations in the 6-month interim pathology report suggests that an exposure duration >6 months may be required before RDX-induced effects on the urogenital system are observed. Suppurative prostatitis was observed with increasing incidence in each dose group in the study at 24 months. Considered as a group, treatment-related kidney and urogenital lesions may have led to a blockage that resulted in urinary stasis. Reduced urinary flow and/or retrograde flow may have contributed to an environment that allowed bacterial infection of the prostate. Thus, while an opportunistic bacterial infection could be the proximal cause of the suppurative prostatitis (ATSDR, 2012; Sweeney et al., 2012a; Crouse et al., 2006), it may have been secondary to the effects of RDX on the urogenital system. This hypothesis is consistent with the observed dose-related increase in incidence of suppurative prostatitis.

Although the ultimate sequence of effects in the urogenital system is unclear, even from review of the scheduled sacrifices at 6 or 12 months on study, it is plausible that suppurative prostatitis would occur after other kidney or bladder lesions that resulted in the initial blockage and urinary stasis. The incidence of suppurative prostatitis reported in Levine et al. (1983b) was increased at doses lower than the doses associated with an increased incidence of other urogenital lesions. However, the incidence of bladder lesions may have been underreported, as the bladders were only examined following observation of a gross abnormality. Bladder distension was reported sporadically among the lower dose groups (0.3, 1.5, or 8.0 mg/kg-day), but the bladder was not routinely examined in these groups (Levine et al., 1983b). Although the pathogenesis of kidney and urogenital system effects cannot be established, the available evidence is consistent with suppurative prostatitis as an indirect effect of RDX exposure and as a marker for the broader array of kidney and urogenital system effects observed by Levine et al. (1983b).

Changes in kidney weights in subchronic oral toxicity studies in rats, dogs, and monkeys did not show a clear pattern of increase or decrease associated with RDX exposure. Kidney weight changes were either not dose-related or were inconsistent across sexes when absolute and relative weights were compared (see Table 1-5). Less weight is placed on evidence of organ weight changes from chronic (2-year) studies (Lish et al., 1984; Hart, 1976) because normal physiological changes

associated with aging and intercurrent disease may contribute to inter-animal variability that could confound organ weight interpretation (Sellers et al., 2007).

Exposure to HMX, the major contaminant in many of the available RDX studies, was associated with histopathological changes in the kidney and alterations in renal function in female, but not male, rats fed doses ≥450 mg/kg-day HMX for 13 weeks (see the Integrated Risk Information System [IRIS] assessment of octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine [HMX] at http://www.epa.gov/iris). No effects were observed at doses ≤115 mg/kg-day. Because the percentage of HMX as an impurity ranged from 3 to 10%, resulting in HMX exposures ≤60 mg/kg-day in the studies of RDX toxicity, the contribution of HMX to the observed kidney toxicity in studies of RDX is expected to be negligible. Further, differences in the pattern of toxicity (i.e., kidney effects observed only in RDX-exposed males and HMX-exposed females) also suggest that HMX contaminants were not responsible for kidney effects in rats exposed to RDX.

Table 1-4. Evidence pertaining to kidney effects in humans

Reference and study design	Results		
Hathaway and Buck (1977) Cross-sectional study, 2,022 workers, 1,491 participated (74% response rate). Analysis group: limited to whites; 69 workers exposed to RDX alone and 24 workers exposed to RDX and HMX, compared to 338 workers not exposed to RDX, HMX, or TNT. **Exposure measures**: Exposure determination based on job title and industrial hygiene evaluation; exposed subjects assigned to two groups: undetected (<LOD) or ≥0.01 mg/m^3 (mean for employees with exposures ≥LOD: 0.28 mg/m^3). **Effect measures**: Renal function tests (blood) **Analysis**: Types of statistical tests were not reported (assumed to be t-tests for comparison of means and χ^2 tests for comparison of proportions).	**Renal function tests: mean** *(standard deviation not reported)*		
		RDX exposed males*	
Test	Referent (n = 237)	Undetected (<LOD) (n = 22)	>0.01 mg/m^3 (n = 45)
BUN	15.5	15.6	16.4
Total protein	7.2	7.2	7.3
		RDX exposed females*	
	Referent (n = 101)	Undetected (<LOD) (n = 1)	>0.01 mg/m^3 (n = 25)
BUN	13.2	8	12.6
Total protein	7.3	7.6	7.2
	*Includes both workers exposed to RDX alone and RDX and HMX. No differences were statistically significant in men or women.		

LOD = limit of detection

Table 1-5. Evidence pertaining to kidney and other urogenital system effects in animals

Reference and study design	Results					
Histopathological lesions						
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2−98.7% pure, with 3−10% HMX as contaminant; 83−89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	The incidence of cytoplasmic vacuolization of renal tubules was greater for RDX-treated males than the control group males after 6 mo of treatment. However, at 12 and 24 mo of treatment, this lesion was observed as frequently in controls as males treated with RDX. There was no increase in incidence of this lesion in females at any time point.					
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Histopathological examination of kidney did not reveal any significant differences compared to controls; lesions observed were not attributed to RDX treatment; incidence data were reported only for control and 10 mg/kg-d groups.					
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2−98.7% pure, with 3−10% HMX as contaminant; 83−89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs Note: More detailed histopathological results, including interim sacrifice data at 6 and 12 mo, are provided in Tables 1-6 to 1-8.	Data for male rats sacrificed on schedule (SS) and those that died spontaneously or were sacrificed moribund (SDMS) (summarized below) were analyzed separately; incidence data were not reported for females.					
	Doses	0	0.3	1.5	8.0	40
	Kidney, medullary papillary necrosis; 24 mo *(incidence)*					
	(SS)	0/38	0/36	0/25	0/29	0/4
	(SDMS)	0/17	1/19	0/27	0/26	18/27*
	(Sum)	0/55	1/55	0/52	0/55	18/31*
	Kidney, suppurative pyelitis; 24 mo *(incidence)*					
	(SS)	0/38	0/36	0/25	0/29	0/4
	(SDMS)	0/17	1/19	0/27	1/26	5/27*
	(Sum)	0/55	1/55	0/52	1/55	5/31*
	Kidney, uremic mineralization; 24 mo *(incidence)*					
	(SS)	1/38	0/36	0/25	0/29	0/4
	(SDMS)	0/17	1/19	2/27	0/26	13/27
	(Sum)	1/55	1/55	2/52	0/55	13/31

Reference and study design	Results					
	Urinary bladder, luminal distention; 24 mo *(incidence)*					
	(SS)	0/38	0/36	0/25	0/29	1/4*
	(SDMS)	0/16	2/19	1/27	3/22	24/28*
	(Sum)	0/54	2/55	1/52	3/51	25/32*
	Urinary bladder, cystitis hemorrhagic/suppurative; 24 mo *(incidence)*					
	(SS)	0/38	0/36	0/25	1/29	0/4
	(SDMS)	0/16	2/19	1/27	0/22	18/27*
	(Sum)	0/54	2/55	1/52	1/51	18/31*
	Prostate, suppurative inflammation (prostatitis); 24 mo *(incidence)*					
	SS	0/38	1/36	2/25*	4/29*	0/4
	SDMS	2/16	3/19	7/27*	8/26	19/27*
	(Sum)	2/54	4/55	9/52*	12/55*	19/31*
Cholakis et al. (1980) Mice, B6C3F$_1$, 10−12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 80, 60, 40 mg/kg-d for 2 wks followed by 0, 80, 160, or 320 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Doses	0	80		160	320
	Tubular nephrosis *(incidence)*					
	M	0/10	–		–	4/9*
	F	0/11	–		–	1/11
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Histopathological examination of kidney did not reveal any significant differences compared to controls; incidence data were reported only for control and 40 mg/kg-d groups.					

Reference and study design	Results					
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26/sex/group; F2: 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	Data were reported only for F2 generation controls and 5 and 16 mg/kg-d groups.					
	Doses	0	5	16	50	
	Cortical cysts *(incidence)*					
	M	4/10	4/10	8/10	–	
	F	3/10	4/10	8/10	–	
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Prostate, mild subacute inflammation *(incidence)*						
	M	0/10	–	–	–	–	1/8
	Histopathological examination of kidney did not reveal any significant differences compared to controls; incidence data were reported only for control and 15 mg/kg-d groups.						
(Levine et al. (1990); Levine et al. (1981a), 1981b))[b] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Histopathological examination of kidney did not reveal any significant differences compared to controls. No histopathology findings available for the 300 or 600 mg/kg-d dose groups because all rats in these groups died before the 13-wk necropsy.					
Hart (1974) Dogs, Beagle, 3/sex/group Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	Histopathological examination of kidney did not reveal any significant differences compared to controls; incidences were reported only for control and 10 mg/kg-d groups.					
Martin and Hart (1974) Monkeys, Cynomolgus or Rhesus[c], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	Doses	0	0.1	1	10	
	Medulla; mineralization, minimal to mild *(incidence)*					
	M + F	0/6	1/6	0/6	4/6	

Reference and study design	Results					
Kidney weight[d]						
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100
	Absolute kidney weight at 104 wks *(percent change compared to control)*					
	M	0%	−1%	4%	9%*	19%*
	F	0%	3%	1%	1%	−2%
	Relative kidney weight at 104 wks *(percent change compared to control)*					
	M	0%	3%	6%	11%*	27%*
	F	0%	1%	1%	2%	19%*
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Doses	0	1.0	3.1	10	
	Absolute kidney weight *(percent change compared to control)*					
	M	0%	−3%	−7%	2%	
	F	0%	14%	−4%	8%	
	Relative kidney weight *(percent change compared to control)*					
	M	0%	−1%	−4%	4%	
	F	0%	22%	3%	18%	
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Absolute kidney weight at 105 wks *(percent change compared to control)*					
	M	0%	2%	−7%	1%	0%
	F	0%	3%	3%	2%	2%
	Relative kidney weight at 105 wks *(percent change compared to control)*					
	M	0%	1%	0%	2%	20%*
	F	0%	3%	6%	5%	21%*
Cholakis et al. (1980) Mice, B6C3F$_1$, 10–12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size Experiment 1: 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute kidney weight *(percent change compared to control)*						
	M	0%	–	–	–	18%	2%
	F	0%	–	–	–	−8%	−5%
	Relative kidney weight *(percent change compared to control)*						
	M	0%	–	–	–	29%	0%
	F	0%	–	–	–	−8%	−3%

Reference and study design	Results						
Experiment 2: 0, 40, 60, or 80 mg/kg-d for 2 wks followed by 0, 320, 160, or 80 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Doses	0		80	160		320
	Absolute kidney weight *(percent change compared to control)*						
	M	0%		8%	11%		13%
	F	0%		−5%	−3%		0%
	Relative kidney weight *(percent change compared to control)*						
	M	0%		5%	9%		10%
	F	0%		−5%	−4%		−5%
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute kidney weight *(percent change compared to control)*						
	M	0%	–	–	–	−2%	−5%
	F	0%	–	–	–	1%	0%
	Relative kidney weight *(percent change compared to control)*						
	M	0%	–	–	–	1%	5%
	F	0%	–	–	–	6%	6%
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26/sex/group; F2: 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	Doses	0		5	16		50
	Absolute kidney weight *(percent change compared to control)*						
	M	0%		6%	−12%		–
	F	0%		−4%	−21%*		–
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Absolute kidney weight *(percent change compared to control)*						
	M	0%	−3%	−4%	−1%	3%	5%
	F	0%	2%	5%	13%*	10%	15%*
	Relative kidney weight *(percent change compared to control)*						
	M	0%	3%	6%	2%	1%	3%
	F	0%	1%	−3%	−1%	−6%	−7%*

Reference and study design	Results						
(Levine et al. (1990); Levine et al. (1981a), 1981b))[b] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Doses	0	10	30	100	300	600
	Absolute kidney weight *(percent change compared to control)*						
	M	0%	1%	1%	−9%	–	–
	F	0%	1%	3%	−1%	–	–
	Relative kidney weight *(percent change compared to control)*						
	M	0%	5%	7%	10%	–	–
	F	0%	3%	5%	2%	–	–
Hart (1974)[e] Dogs, Beagle, 3/sex/group Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	Doses	0		0.1	1		10
	Absolute kidney weight *(percent change compared to control)*						
	M	0%		–	–		38%
	F	0%		–	–		−18%
Martin and Hart (1974)[e] Monkeys, Cynomolgus or Rhesus[c], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	Doses	0		0.1	1		10
	Absolute kidney weight *(percent change compared to control)*						
	M + F	0%		−2%	−3%		4%

*Statistically significant ($p < 0.05$) based on analysis by study authors.
[a]Doses were calculated by the study authors.
[b]Levine et al. (1981a) is a laboratory report of a 13-week study of RDX in F344 rats; two subsequently published papers (Levine et al., 1990; Levine et al., 1981b) present subsets of the data provided in the full laboratory report.
[c]The species of monkey used in this study was inconsistently reported in the study as either Cynomolgus (in the methods section) or Rhesus (in the summary).
[d]An analysis by Craig et al. (2014) found a statistically significant correlation between absolute, but not relative, kidney weights and renal histopathology. Therefore, only absolute kidney weight data from RDX studies are presented in Figure 1-2.
[e]Kidney weight data from the Hart (1974) and Martin and Hart (1974) studies were considered less informative than other studies. Hart (1974) reported organ weight data for high-dose dogs (3/sex/group) only, and the kidney weights from Martin and Hart (1974) were highly variable across monkeys (e.g., kidney weights for the control animals ranged from 4.9 to 13.1 g). Therefore, kidney weight data from these two studies were not presented in the exposure-response array for kidney and other urogenital system effects (Figure 1-2).

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.
SDMS -- spontaneous death or moribund sacrifice; SS -- scheduled sacrifice

Table 1-6. Six-, 12-, and 24-month incidence of kidney endpoints in male F344 rats reported for statistical evaluation in Levine et al. (1983b)

Doses (mg/kg-d)	0	0.3	1.5	8.0	40
Medullary papillary necrosis *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	–	–	0/5
Sum	0/10	0/10	0/10	0/10	0/15
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	15/19*
Sum	0/10	0/10	0/13	0/10	15/29*
24 mo					
SS	0/38	0/36	0/25	0/29	0/4
SDMS	0/17	1/19	0/27	0/26	18/27*
Sum	0/55	1/55	0/52	0/55	18/31*
Pyelitis *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	–	–	0/5
Sum	0/10	0/10	0/10	0/10	0/15
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	1/19
Sum	0/10	0/10	0/13	0/10	1/29
24 mo					
SS	0/38	0/36	0/25	0/29	0/4
SDMS	0/17	1/19	0/27	1/26	5/27*
Sum	0/55	1/55	0/52	1/55	5/31*
Pyelonephritis *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	–	–	0/5
Sum	0/10	0/10	0/10	0/10	0/15

Doses (mg/kg-d)	0	0.3	1.5	8.0	40
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	1/19
Sum	0/10	0/10	0/13	0/10	1/29
24 mo					
SS	0/38	0/36	0/25	1/29	0/4
SDMS	0/17	0/19	2/27	1/26	1/27
Sum	0/55	0/55	2/52	2/55	1/31

*Statistically significant ($p < 0.05$) based on analysis by study authors.

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.
SDMS -- spontaneous death or moribund sacrifice; SS -- scheduled sacrifice

Source: Levine et al. (1983b).

Table 1-7. Six-, 12-, and 24-month incidence of urinary bladder endpoints in male F344 rats reported for statistical evaluation in Levine et al. (1983b)

Doses (mg/kg-d)	0	0.3	1.5	8.0	40
Luminal distention *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	–	–	0/5
Sum	0/10	0/10	0/10	0/10	0/15
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	18/19*
Sum	0/10	0/10	0/13	0/10	18/29
24 mo					
SS	0/38	0/36	0/25	0/29	1/4*
SDMS	0/16	2/19	1/27	3/22	24/28*
Sum	0/54	2/55	1/52	3/51	25/32*
Cystitis, hemorrhagic/suppurative *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	–	–	0/5

Doses (mg/kg-d)	0	0.3	1.5	8.0	40
Sum	0/10	0/10	0/10	0/10	0/15
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	17/19*
Sum	0/10	0/10	0/13	0/10	17/29
24 mo					
SS	0/38	0/36	0/25	1/29	0/4
SDMS	0/16	2/19	1/27	0/22	18/27*
Sum	0/54	2/55	1/52	1/51	18/31*

*Statistically significant ($p < 0.05$) based on analysis by study authors.

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.
SDMS -- spontaneous death or moribund sacrifice; SS -- scheduled sacrifice

Source: Levine et al. (1983b).

Table 1-8. Six-, 12-, and 24-month incidence of prostate endpoints in male F344 rats reported for statistical evaluation in Levine et al. (1983b)

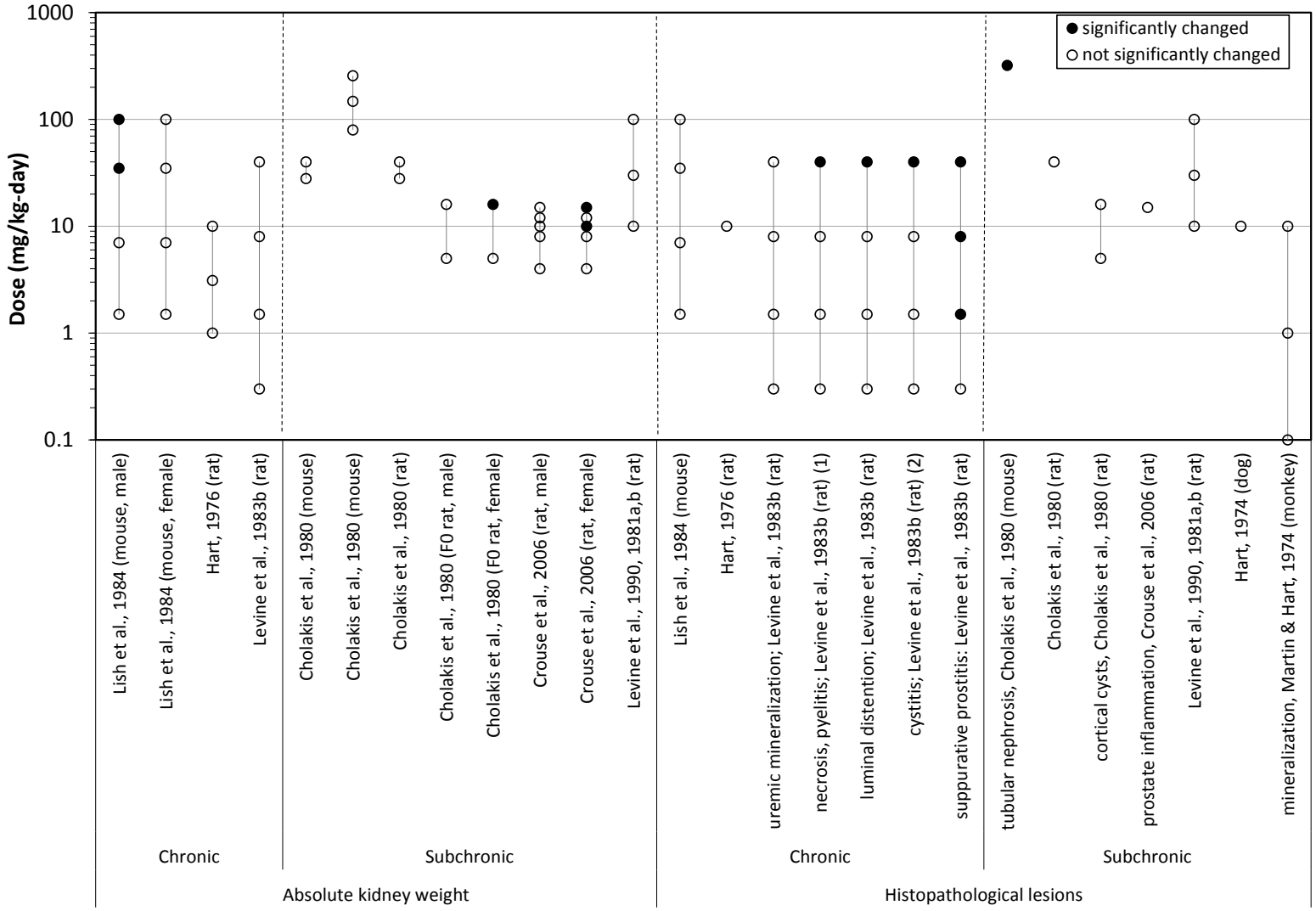
Doses (mg/kg-d)	0	0.3	1.5	8.0	40
Spermatic granuloma *(incidence)*					
6 mo					
SS	0/10	2/10	2/10	1/10	6/10*
SDMS	–	–	–	–	2/5
Sum	0/10	2/10	2/10	1/10	8/15*
12 mo					
SS	0/10	0/10	1/10	1/10	0/10
SDMS	–	–	0/3	–	0/19
Sum	0/10	0/10	1/13	1/10	0/29
24 mo					
SS	0/38	0/36	0/25	0/29	0/4
SDMS	0/16	0/19	0/27	0/26	0/27
Sum	0/54	0/55	0/52	0/55	0/31
Suppurative inflammation *(incidence)*					
6 mo					
SS	0/10	0/10	0/10	0/10	0/10

Doses (mg/kg-d)	0	0.3	1.5	8.0	40
SDMS	–	–	–	–	0/5
Sum	0/10	0/10	0/10	0/10	0/15
12 mo					
SS	0/10	0/10	0/10	0/10	0/10
SDMS	–	–	0/3	–	0/19
Sum	0/10	0/10	0/13	0/10	0/29
24 mo					
SS	0/38	1/36	2/25*	4/29*	0/4
SDMS	2/16	3/19	7/27*	8/26	19/27*
Sum	2/54	4/55	9/52*	12/55*	19/31*

*Statistically significant ($p < 0.05$) based on analysis by study authors.

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.
SDMS -- spontaneous death or moribund sacrifice; SS -- scheduled sacrifice

Source: Levine et al. (1983b).



Note: Filled circle indicates that response was statistically significantly different from the control.
(1) Statistical significance determined from incidence at time of scheduled sacrfice. (2) Statistical significance determined from incidence at spontaneous death.

Figure 1-2. Exposure-response array of kidney and other urogenital system effects.

Mechanistic Evidence

No MOA information is available for RDX-induced kidney and other urogenital effects. However, mechanistic information underlying the neurotoxicity observed with RDX exposure, and the specific affinity of RDX to the $GABA_A$ receptor-convulsant site (Williams et al., 2011; Williams and Bannon, 2009), suggests a biologically plausible role for the $GABA_A$ receptor in RDX-related effects on the urogenital system, and provides some potential MOA hypotheses for the effects reported in Levine et al. (1983b).

One hypothesis is that urogenital effects of RDX are caused by interactions with $GABA_A$ receptors mediating inputs to the urogenital system. GABA and GABA receptors have been identified in a number of peripheral tissues (Erdö et al., 1991; Ong and Kerr, 1990; Erdo, 1985). Brar et al. (2014) demonstrated that pretreatment with picrotoxin reduced the renoprotective effects of sodium valproate (which acts on both $GABA_A$ and $GABA_B$ receptors) in a rat model of ischemia-induced acute kidney injury, suggesting that $GABA_A$ receptors may be important in renal function. GABA is believed to play a role in the regulation of urination and bladder capacity (reviewed in Fowler et al. (2008) and Yoshimura and de Groat (1997)). In rats, injection of a $GABA_A$ receptor agonist inhibits the urination reflex (Igawa et al., 1993; Kontani et al., 1987). $GABA_A$ agonists injected into the periaqueductal gray area in rats inhibited reflex bladder activity, while injection of an antagonist reduced bladder capacity and increased the frequency of bladder reflex activity (Stone et al., 2011). RDX would be expected to act like an antagonist and increase bladder activity (which would not result in urinary stasis), although the impact of chronic exposure to RDX acting as a $GABA_A$ receptor antagonist is not known. Evidence of GABAergic signaling regulating bladder function, and the hypothesized disruption of that regulation by RDX via interaction with $GABA_A$ receptors, may plausibly account for the kidney and other urogenital lesions, including suppurative prostatitis, observed by Levine et al. (1983b); however, no evidence to support this hypothesized MOA is available.

Other potential mechanisms by which RDX, through $GABA_A$ binding, may lead to kidney and urogenital effects are less apparent. Alterations in hormonal signaling or circulating levels of estrogen or prolactin may lead to prostatitis. Prostate inflammation has been associated with endocrine disruptors in the environment (Cowin et al., 2010), and increased prolactin has been shown to cause lateral lobe prostatitis (Stoker et al., 1999b; Stoker et al., 1999a; Tangbanluekal and Robinette, 1993; Robinette, 1988). Typically, the inflammation seen is chronic and does not reverse over time (Robinette, 1988). Functional $GABA_A$ receptors have been identified in the anterior pituitary (Zemkova et al., 2008; Mayerhofer et al., 2001), which also serves as the primary source of prolactin. Thus, the prostate inflammation observed in the rat in the 2-year study by Levine et al. (1983b) could have been produced by disruption of pituitary prolactin or another hormonal signal via interference with normal regulatory GABA-related hormonal control. However, no direct evidence for this hypothesized MOA is available. Levine et al. (1983b) did not evaluate serum endocrine measures or pituitary weights, and pituitary adenomas that could account for higher

prolactin levels were not observed. A MOA hypothesis based on pituitary-mediated alterations in endocrine signaling also does not explain the other urogenital lesions observed by Levine et al. (1983b).

Another hypothesis is that the prostate effects could be mediated through an autoimmune inflammatory response. $GABA_A$ receptor transcripts have been identified in immune cells of mouse models (Reyes-García et al., 2007; Tian et al., 2004), and $GABA_A$ receptor agonists have decreased cytotoxic immune responses and hypersensitivity reactions (Tian et al., 1999; Bergeret et al., 1998). In a mouse autoimmune model of multiple sclerosis, Bhat et al. (2010) found that treatment of macrophages challenged with lipopolysaccharide with various GABA agonists decreased cytokine production; addition of picrotoxin (which may have effects similar to those of RDX, as it binds to the same site) was able to reduce this effect. However, picrotoxin on its own did not significantly alter cytokine production, suggesting that the effects are limited to reversal of agonist-induced GABAergic activity (Johnson, 2015b). If an autoimmune mechanism was contributing to the effects observed with RDX exposure, it is unclear why inflammation would be limited to the prostate. RDX has also tested negative in the only battery of immunotoxicity tests to which it was subjected (Crouse et al., 2006).

If the urogenital effects are mediated through localized interaction with $GABA_A$ receptors, another possibility would be that that effects would result from direct interactions with $GABA_A$ receptors located on the prostate. $GABA_A$ receptors have been identified on the prostate (Napoleone et al., 1990), providing a potential mechanism by which RDX could interact directly with the prostate. However, this would require that the prostate is actively maintained in a non-inflamed state, mediated by GABA; RDX binding to $GABA_A$ receptor-convulsant sites on the prostate would result in a reduction of the inhibitory effects of the GABA receptor, leading to increased inflammation (Johnson, 2015b). No evidence was found to support this potential pathway leading to prostate inflammation.

In summary, there are no studies available that inform mechanistically how RDX might lead to kidney and other urogenital effects. There is evidence that RDX binds to $GABA_A$ receptors in neuronal tissues (Williams et al., 2011; Williams and Bannon, 2009), and it is biologically plausible that binding to the GABA receptor could occur in other tissues as well, contributing to the observed kidney and urogenital effects. Among the mechanistic information presented above, MOAs that require direct action on the prostate are considered less likely because the available information suggests that the prostatitis is a secondary effect. However, the ways that $GABA_A$ receptors work in non-neuronal tissues and organs is still not well understood, and the MOA by which RDX induces kidney and other urogenital effects is unknown.

Integration of Kidney and Urogenital System Effects

Evidence for kidney effects resulting from RDX exposure consists of human case reports and findings of histopathological changes in rodents. In humans, evidence for kidney effects (including decreased urine output, blood in urine, and proteinuria) is limited to individuals with acute

accidental exposure (ingestion and inhalation) to unknown amounts of RDX. No RDX-related changes in kidney parameters were found in a small cross-sectional study of RDX-exposed workers (Hathaway and Buck, 1977).

A dose-related increase in the incidence of suppurative prostatitis in male rats (Levine et al., 1983b) provides the strongest evidence of RDX-associated kidney and other urogenital system effects. As discussed above, the incidence of suppurative prostatitis is considered to be an indicator for the broader array of kidney and other urogenital effects seen in this study. Levine et al. (1983b) identified other histopathological effects (papillary necrosis, pyelitis, luminal distension, and cystitis) in the kidney and bladder, but at the highest dose only. A second 2-year study in Sprague-Dawley rats found no histopathological changes in the kidney or urogenital system (Hart, 1976), but exposure levels used in this study were low compared to Levine et al. (1983b). Other measures of kidney effects, specifically kidney weights and serum chemistry parameters, did not provide consistent evidence of dose-related changes associated with RDX exposure. In light of the dose-related increase in suppurative prostatitis and the lack of support for an alternative (i.e., non-RDX-related) basis for this effect, kidney and urogenital effects are a potential human hazard of RDX exposure.

1.2.3. Reproductive and Developmental Effects

No human studies were identified that evaluate the potential of RDX to cause reproductive or developmental effects. Information relevant to an examination of the association between RDX exposure and reproductive and developmental effects comes from a 2-generation reproductive toxicity study in rats and developmental studies in rats and rabbits involving oral administration of RDX during gestation. In addition, oral subchronic and chronic studies in experimental animals provide information useful for examining the association between RDX exposure and effects specifically on the male reproductive system. A summary of the reproductive and developmental effects associated with RDX exposure is presented in Tables 1-9 and 1-10 and Figures 1-3 and 1-4. Studies are ordered in the evidence tables and exposure-response arrays by duration of exposure and then by species.

Reproductive Effects

Evidence of male reproductive toxicity is provided by the finding of testicular degeneration in male mice. An increased incidence of testicular degeneration (10–11%) was observed in male B6C3F$_1$ mice exposed to ≥35 mg/kg-day RDX for 2 years in the diet compared to concurrent (0%) and historical (1.5%) controls (Lish et al., 1984). Reductions in absolute testicular weight were observed, but the magnitude of this effect was small (≤6% compared to controls) and not dose-related. An increased incidence of germ cell degeneration was observed in rats exposed to 40 mg/kg-day (40%) compared with controls at 12 months (0%); by 24 months, almost all male rats (including controls) had testicular masses (interstitial cell tumors), and no instances of germ cell degeneration were identified in control or RDX-treated groups (Levine et al., 1983b). No dose-

related histopathological changes in the testes were identified in other studies in rats (Crouse et al., 2006; Levine et al., 1990; Levine et al., 1981a, b; Hart, 1976) or dogs (Hart, 1974). Changes in testicular weight were inconsistent across studies, with an equivalent number of studies identifying decreases (Crouse et al., 2006; Lish et al., 1984; Cholakis et al., 1980) or increases (Levine et al., 1990; Levine et al., 1981a, b; Cholakis et al., 1980; Hart, 1976, 1974) in testicular weight; in most cases, the changes in testicular weight were small (≤10% change compared to control) and not dose-related.

Reproductive function was assessed in two separate studies reported by Cholakis et al. (1980). No specific effects on reproductive function were observed in F0 and F1 CD rats exposed to ≤16 mg/kg-day RDX in the Cholakis et al. (1980) two-generation study. The highest dose tested, 50 mg/kg-day, was associated with reductions in fertility (specifically a decreased number of pregnancies) in the F0 generation, although these changes were not statistically significant. The finding of lower fertility rates only at the 50 mg/kg-day dose, a dose associated with reduced body weight and feed consumption and increased mortality (9% in male rats and 27% in female rats), suggests that effects on reproductive function were likely due to the general toxicity of RDX rather than a direct effect of RDX on reproduction. In the dominant lethal mutation study, which used the F0 males from the two-generation reproductive toxicity study, no effects on fertility were observed in male rats exposed to ≤16 mg/kg-day RDX. Pregnancy rates were lower in untreated females mated to males exposed to 50 mg/kg-day RDX for 15 weeks prior to mating; the authors attributed this effect to a treatment-related decrease in the well-being of the males in this high-dose group (Cholakis et al., 1980).

Developmental Effects

Animal studies have reported decreases in offspring survival following administration of RDX. Pup survival rates in the F0 and F1 generations (including both stillborn pups and postnatal deaths through the age of weaning) were statistically significantly decreased in RDX-exposed CD rats compared to controls in the only available two-generation reproductive toxicity study of RDX (Cholakis et al., 1980). This observation was noted only at the highest dose tested (50 mg/kg-day) that also produced toxicity in adults (mortality [18%], reduced body weights [8–14%], and reduced food consumption [10–17%]). Decreased fetal viability was observed at the highest dose tested, 20 mg/kg-day, in a developmental toxicity study in F344 rats (Cholakis et al., 1980), although no effect on live fetuses was observed in a developmental toxicity study in Sprague-Dawley rats at the same dose (Angerhofer et al., 1986); both of these studies reported significant mortality (29–31%) in dams at 20 mg/kg-day. Increased resorptions were similarly limited to the highest dose tested (20 mg/kg-day) (Cholakis et al., 1980). Both of these studies started treatment with RDX on gestational day (GD) 6, which may contribute to the incidence of resorptions observed in the control and treated groups. As noted in EPA's *Guidelines for Developmental Toxicity Risk Assessment* (U.S. EPA, 1991), treatment beginning around the time of implantation may result in an increase in implantation loss that reflects variability that is not treatment related. There was no evidence of

maternal toxicity, embryotoxicity, or decreased fetal viability in a teratology study of pregnant New Zealand White (NZW) rabbits administered RDX by gavage from GD 7 to 29 at doses up to 20 mg/kg-day (Cholakis et al., 1980), suggesting that rabbits may be less sensitive to RDX toxicity than rats.

Statistically significant, dose-related reductions in fetal body weight and length were reported in Sprague-Dawley rats administered RDX by gavage from GD 6 to 15 (Angerhofer et al., 1986).[9] Decreased fetal body weight (9%) and body length (5%), with statistically significant trends, were observed at 20 mg/kg-day, a dose that produced significant (31%) mortality in the dams. A similar reduction in fetal body weight of 7% (not statistically significant) was observed in F344 rats exposed to RDX at 20 mg/kg-day, a dose associated with 29% maternal mortality (Cholakis et al., 1980). Dose-related reductions in fetal body weight were not observed in NZW rabbits at doses up to 20 mg/kg-day (Cholakis et al., 1980).

No treatment-related effects on morphological development have been reported in rats exposed to a dose as high as 20 mg/kg-day RDX, a dose that resulted in 29–31% maternal mortality (Angerhofer et al., 1986; Cholakis et al., 1980). Examination of rabbits administered RDX at doses up to 20 mg/kg-day from GD 7 to 29 also provided no evidence of treatment-related developmental anomalies (Cholakis et al., 1980). Although increased incidences of enlarged frontal fontanel and unossified sternebrae were observed in fetuses of all groups of NZW rabbits administered RDX (Cholakis et al., 1980), these developmental anomalies did not exhibit a dose-related increase in the number of either fetuses or litters affected, and were thus interpreted as not being treatment-related by the study authors (Cholakis et al., 1980). This interpretation is supported by the following additional considerations. Neither individual litter data nor historical control data from the performing laboratory were available to assist in the interpretation of these findings. A report of historical control incidences of fetal skeletal observations in NZW rabbits for 224 prenatal developmental toxicology studies conducted in 8 contract research laboratories during the period of 1988–1992 (MTA, 1992) included findings from 26,166 fetuses of 3,635 litters. Background control incidences of enlarged anterior fontanel were observed in 8 fetuses (0.031%) of 7 litters (0.193%), while sternebrae agenesis (which may not be entirely comparable to the finding of unossified sternebrae in Cholakis et al. (1980) was found in 10 fetuses (0.038%) of 5 litters (0.138%). Although the use of concurrent control data is preferable for the interpretation of developmental toxicity data, this historical information supports the low control incidences of these findings in the Cholakis et al. (1980) study as being within typical historical parameters. It is also noted that the non-dose-related pattern of increased enlarged fontanel and unossified sternebrae across treated groups in Cholakis et al. (1980) was similar to the pattern of decreases in fetal body

[9]The statistical analyses presented by the study authors were performed on a per fetus basis; EPA's *Guidelines for Developmental Toxicity Risk Assessment* (U.S. EPA, 1991) recommend that fetal data be analyzed on a per litter (rather than per fetus) basis. In a reanalysis of the Angerhofer et al. (1986) data by EPA on a per litter basis, fetal body weight and length showed statistically significant decreasing trends.

weight in the same study, suggesting a possible link between these particular sternebral and fontanel anomalies with fetal growth status. Given the lack of dose-related increases in the incidences of these anomalies, and patterns that mirrored fetal body weight decreases (which were also not dose-related), the findings of enlarged frontal fontanel and unossified sternebrae were not considered treatment-related. Gestational administration of RDX to NZW rabbits did not result in any other dose- and treatment-related skeletal abnormalities.

Table 1-9. Evidence pertaining to male reproductive effects in animals

Reference and study design	Results					
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100
	Testicular degeneration *(incidence)*					
		0/63	2/60	2/62	6/59	3/27[a]
	Absolute testes weight; wk 105 *(percent change compared to control)*					
		0%	−6%	0%	−2%	−6%
	Relative testes weight; wk 105 *(percent change compared to control)*					
		0%	−4%	2%	−2%	−2%
Hart (1976) Rats, Sprague-Dawley, 100/sex/dose Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Doses	0	1.0	3.1	10	
	Absolute testes (with epididymis) weight; wk 104					
		0%	−2%	2%	5%	
	Relative testes (with epididymis) weight; wk 104					
		0%	−1%	7%	9%	
	Testes were examined microscopically in control and 10 mg/kg-d groups; no degeneration or other treatment-related effects were observed.					
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Testes, germ cell degeneration; 12 mo[b] *(incidence)*					
	SS	0/10	0/10	0/10	0/10	4/10*
	SDMS	–	–	1/3	–	4/19
	Testes, germ cell degeneration; 24 mo *(incidence)*					
	SS	0/38	0/36	0/25	0/29	0/4
	SDMS	0/16	0/19	0/27	0/26	0/27
	Testes weights were not measured at termination due to testicular masses in nearly all males.					

Reference and study design	Results						
Cholakis et al. (1980) Mice, B6C3F$_1$, 10−12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size Experiment 1: 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks Experiment 2: 0, 40, 60, or 80 mg/kg-d for 2 wks followed by 0, 320, 160, or 80 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[c] Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute testes weight *(percent change compared to control)*						
		0%	–	–	–	−4%	−4%
	Relative testes weight *(percent change compared to control)*						
		0%	–	–	–	2%	−1%
	Doses	0		80	160		320
	Absolute testes weight *(percent change compared to control)*						
		0%		4%	−4%		−8%
	Relative testes weight *(percent change compared to control)*						
		0%		1%	−4%		−9%
	Testes were examined microscopically in control and 320 mg/kg-d groups; no effects were observed.						
Cholakis et al. (1980) Rats, F344, 10/sex/dose 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute testes weight *(percent change compared to control)*						
		0%	–	–	–	−2%	0%
	Relative testes weight *(percent change compared to control)*						
		0%	–	–	–	2%	9%
	Testes were examined microscopically in control and 40 mg/kg-d groups; no effects were observed.						
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26/sex/group; F2: 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	In F2 weanling offspring of 0, 5, and 16 mg/kg-d groups. No high-dose F2 animals available.						
	Doses	0		5	16		50
	Absolute testes weight *(percent change compared to control)*						
		0%		3%	−31%		–
	Testes of F2 weanlings were examined microscopically in all groups; no effects were observed.						

Reference and study design	Results						
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Absolute testes weight *(percent change compared to control)*						
		0%	−3%	−5%	−4%	−4%	−8%
	Relative testes weight *(percent change compared to control)*						
		0%	4%	5%	0%	−6%	−10%*
Levine et al. (1981a); Levine et al. (1990); Levine et al. (1981b)[d] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Doses	0	10	30	100	300	600
	Testes, germ cell degeneration *(incidence)*						
		0/10	0/10	0/10	0/10	1/9	1/10
	Absolute testes weight *(percent change compared to control)*						
		0%	1%	1%	−2%	–	–
	Relative testes weight *(percent change compared to control)*						
		0%	4%	5%	19%*	–	–
Hart (1974)[e] Dogs, Beagle, 3/sex/dose Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	Doses	0		0.1	1		10
	Absolute testes (with epididymis) weight *(percent change compared to control)*						
		0%		–	–		51%
	Testes were not examined microscopically.						

*Statistically significant ($p < 0.05$) based on analysis by study authors.

[a]Although the study authors did not observe a statistically significant increase in the incidence of testicular degeneration, they determined that the incidences at the 35 and 175/100 mg/kg-day dose groups were "notable" when compared to concurrent (0%) and historical (1.5%) incidences.

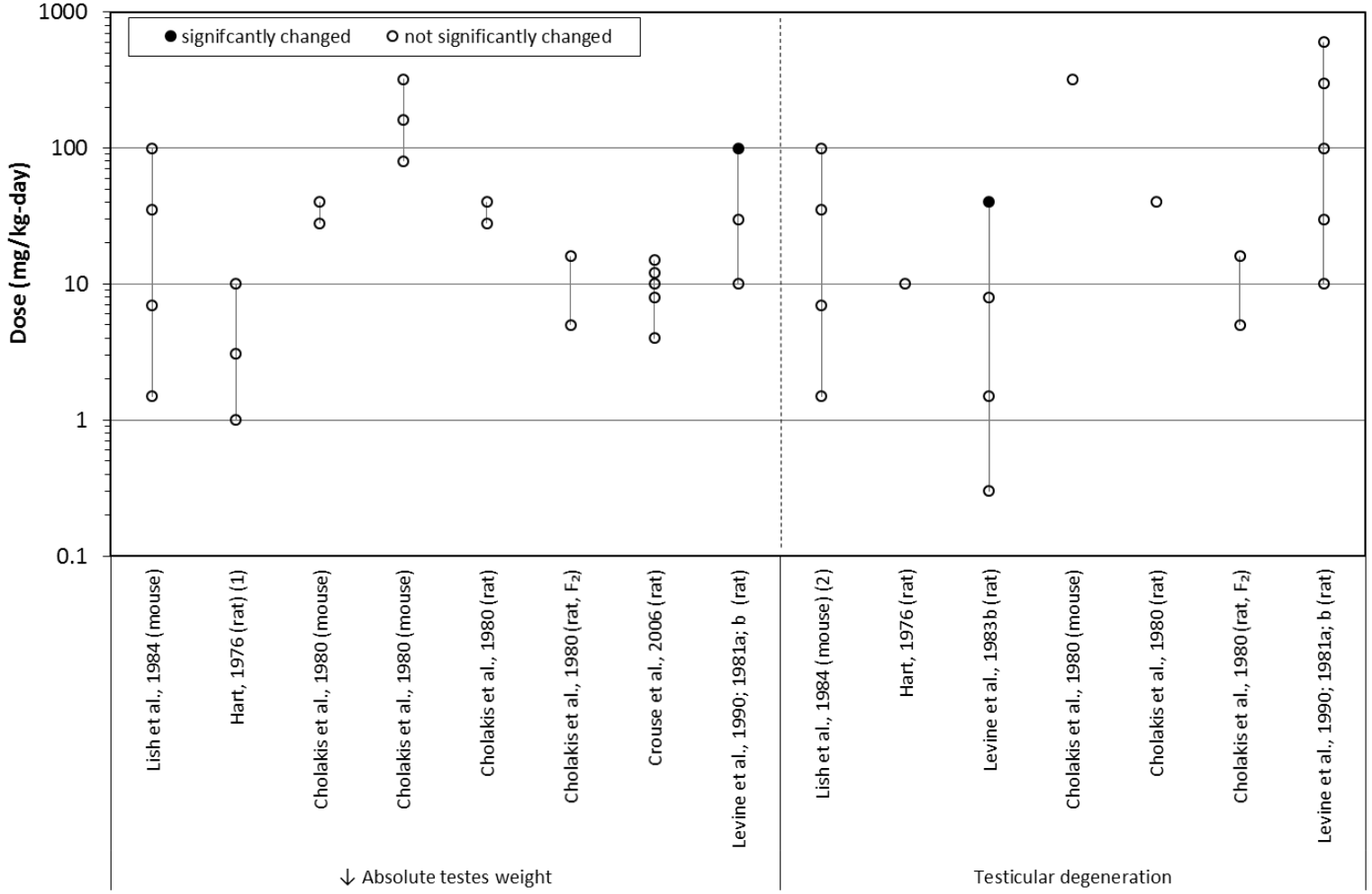
[b]Testicular atrophy was observed at 12 months along with a statistically reduced mean testes weight (compared with controls). By 24 months, almost all male rats (including controls) had testicular masses (interstitial cell tumors); testes weights were not recorded, and an increased incidence of testicular degeneration was not observed.

[c]Doses were calculated by the study authors.

[d]Levine et al. (1981a) is a laboratory report of a 13-week study of RDX in F344 rats; two subsequently published papers (Levine et al., 1990; Levine et al., 1981b) present subsets of the data provided in the full laboratory report.

[e]Because testes weight was reported for only three treated animals in this study, organ data from this study were considered less informative than other studies; therefore, testes weights from Hart (1974) were not presented in the exposure-response array for male reproductive effects.

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.
SDMS -- spontaneous death or moribund sacrifice; SS -- scheduled sacrifice



Note: Filled circle indicates that response was statistically significantly different from the control.
(1) Increased absolute weight of testes and epididymis. (2) Although the study authors did not observe a statistically significant increase in the incidence of testicular degeneration, they determined that the incidences at the 35 and 175/100 mg/kg-day dose groups were "notable" when compared to concurrent (0%) and historical (1.5%) incidences.

Figure 1-3. Exposure response array of male reproductive effects following oral exposure.

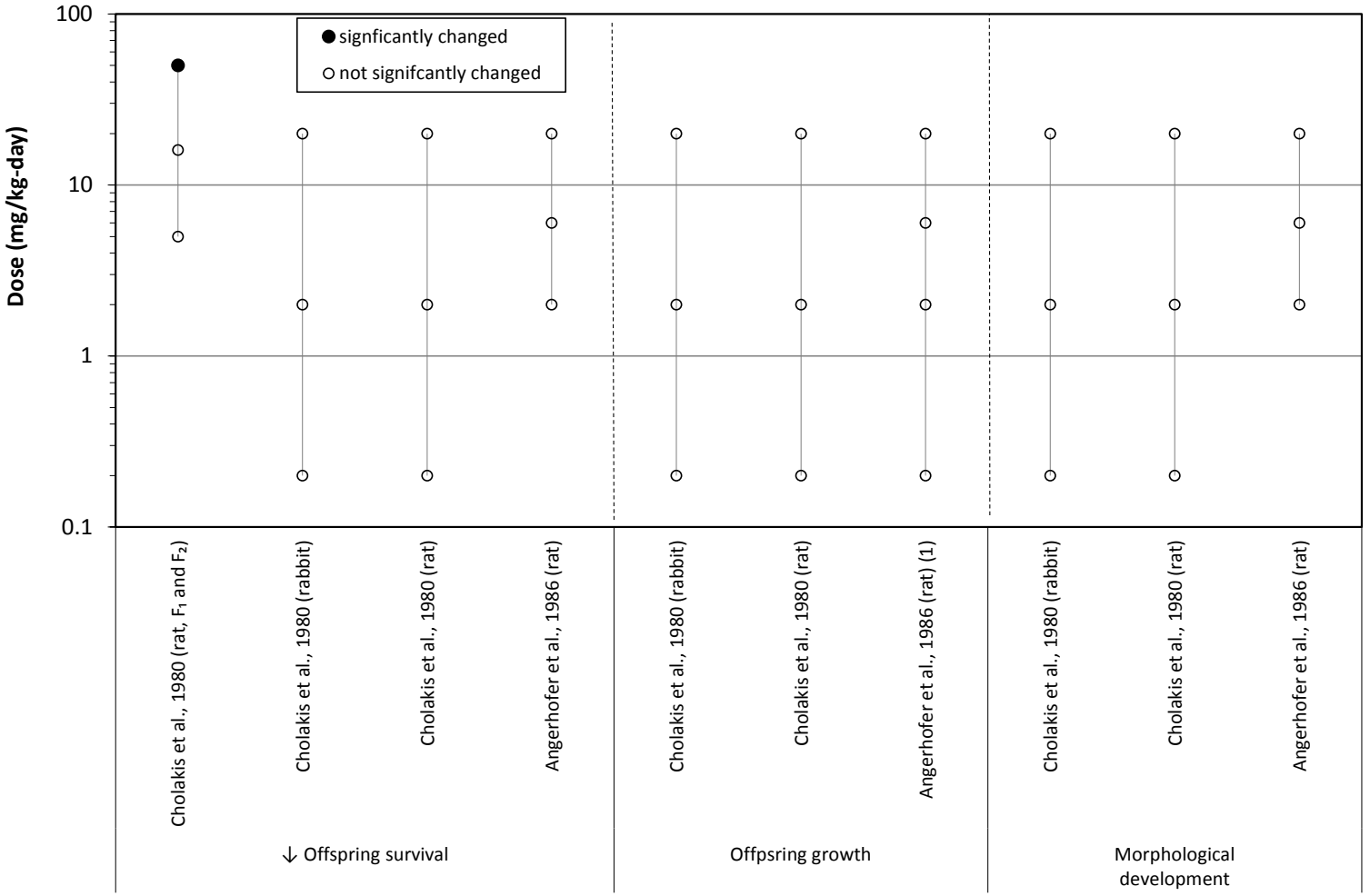
Table 1-10. Evidence pertaining to reproductive and developmental effects in animals

Reference and study design	Results				
Offspring survival					
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26 sex/group; F2: 10 sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	Doses	0	5	16	50
	Stillborn pups *(incidence)*				
	F1	8/207	6/296	4/259	16/92*
	F2	6/288	6/290	2/250	24/46*
	Offspring survival at birth *(percent of fetuses)*				
	F1	96%	98%	98%	83%*
	F2	98%	98%	99%	48%*
	Survival at weaning *(percent of liveborn pups)*				
	F1	87%	96%	90%	8%
	F2	79%	86%	79%	0%
	F0 maternal deaths occurred at 50 mg/kg-d. Only six F1 females in this group survived to serve as parental animals; none of the surviving six died during subsequent treatment. Note: results on a per litter basis were not provided.				
Cholakis et al. (1980) Rabbits, NZW, 11–12/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 7–29	Doses	0	0.2	2	20
	Early resorptions *(mean percent per dam)*				
		6%	5%	4%	1%
	Late resorptions *(mean percent per dam)*				
		8%	5%	3%	3%
	Viable fetuses *(mean percent per dam)*				
		85%	82%	77%	94%
Cholakis et al. (1980) Rats, F344, 24–25 females/group 88.6% pure, with 9% HMX and 2.2% water as contaminants. 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 6–19	Doses	0	0.2	2.0	20
	Early resorptions *(mean percent per dam)*				
		6.0%	2.5%	4.8%	15.3%
	Late resorptions *(mean percent per dam)*				
		0.5%	0.5%	0.3%	1.6%
	Complete litter resorptions *(number of litters)*				
		0	0	0	2
	Viable fetuses *(mean percent per dam)*				
		93.2%	97.6%	94.9%	81.4%
	Significant maternal mortality (7/24 dams) occurred at 20 mg/kg-d.				

Reference and study design	Results				
Angerhofer et al. (1986) Rats, Sprague-Dawley, 39–51 mated females/group (25–29 pregnant dams/group) Purity 90%; 10% HMX and 0.3% acetic acid occurred as contaminants 0, 2, 6, or 20 mg/kg-d Gavage GDs 6–15	Doses	0	2	6	20
	Resorptions *(percent of total implantations)*				
		4.8%	6.1%	5.9%	6.4%
	Early resorptions *(percent of total implantations)*				
		4.8%	6.1%	5.9%	6.2%
	Late resorptions *(percent of total implantations)*				
		0%	0%	0%	0.27%
	Live fetuses *(mean percent per litter)*				
		100%	100%	100%	100%
	Significant maternal mortality (16/51) occurred at 20 mg/kg-d. Percent resorptions and live fetuses based on number of surviving females at time of necropsy.				
Offspring growth					
Cholakis et al. (1980) Rabbits, NZW, 11–12/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 7–29	Doses	0	0.2	2.0	20
	Fetal body weight *(percent change compared to control)*				
		0%	−6.7%	−2.3%	−9.3%
Cholakis et al. (1980) Rats, F344, 24–25 females/group 88.6% pure, with 9% HMX and 2.2% water as contaminants. 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 6–19	Doses	0	0.2	2.0	20
	Fetal body weight *(percent change compared to control)*				
		0%	2%	3%	−7%
	Significant maternal mortality (7/24 dams) occurred at 20 mg/kg-d.				
Angerhofer et al. (1986) Rats, Sprague-Dawley, 39–51 mated females/group (25–29 pregnant dams/group) Purity 90%; 10% HMX and 0.3% acetic acid occurred as contaminants 0, 2, 6, or 20 mg/kg-d Gavage GDs 6–15	Doses	0	2	6	20
	Fetal body weight *(percent change compared to control)*				
		0%	−4%	−2%	−9%[a]
	Fetal body length *(percent change compared to control)*				
		0%	−1%	−1%	−5%[a]
	Significant maternal mortality (16/51) occurred at 20 mg/kg-d.				

Reference and study design	Results				
Morphological development					
Cholakis et al. (1980) Rabbits, NZW, 11–12/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 7–29	Doses	0	0.2	2.0	20
	Spina bifida *(incidence)*				
	Fetuses	0/88	0/99	0/94	3/110
	Litters	0/11	0/11	0/11	2/12
	Misshapen eye bulges *(incidence)*				
	Fetuses	0/88	0/99	0/94	3/110
	Litters	0/11	0/11	0/11	1/12
	Cleft palate *(incidence)*				
	Fetuses	0/39	1/46	2/44	2/52
	Litters	0/11	1/11	1/11	1/12
	Enlarged front fontanel *(incidence)*				
	Fetuses	0/49	5/53	2/50	8/58
	Litters	0/11	2/11	2/11	2/12
	Unossified sternebrae (*incidence*)				
	Fetuses	4/49	12/53	8/50	12/58
	Litters	4/11	7/11	4/11	6/12
Cholakis et al. (1980) Rats, F344, 24–25 females/group 88.6% pure, with 9% HMX and 2.2% water as contaminants. 0, 0.2, 2.0, or 20 mg/kg-d Gavage GDs 6–19	No gross or soft-tissue anomalies were seen in any exposure group. No treatment-related increase in the incidence of litters with skeletal anomalies was observed. Significant maternal mortality (7/24 dams) occurred at 20 mg/kg-d.				
Angerhofer et al. (1986) Rats, Sprague-Dawley, 39–51 mated females/group (25–29 pregnant dams/group) Purity 90%; 10% HMX and 0.3% acetic acid occurred as contaminants 0, 2, 6, or 20 mg/kg-d Gavage GDs 6–15	No treatment-related increase in the incidence of anomalies was observed.				
	Doses	0	2	6	20
	Total malformations *(percent of fetuses with malformations)*				
		1%	1%	0%	2%
	Significant maternal mortality (16/51) occurred at 20 mg/kg-d.				

*Statistically significant ($p < 0.05$) based on analysis by study authors.
[a]Statistically significant dose-related trend ($p < 0.05$) by linear trend test, performed for this assessment. Average fetal weights or lengths for each litter comprised the sample data for this test.



Note: Filled circle indicates that response was statistically significantly different from the control.
(1) Statistically signficant dose-related trend (p <= 0.05) by linear trend test, performed for this assessment.

Figure 1-4. Exposure response array of reproductive and developmental effects following oral exposure.

Integration of Reproductive and Developmental Effects

Testicular effects were reported in male B6C3F$_1$ mice chronically exposed to RDX in the diet for 24 months (Lish et al., 1984). No other studies of equivalent duration were performed in mice to determine the consistency of this effect. Germ cell degeneration was observed in F344 rats at 12 months, but not at 24 months, in a 2-year study (Lish et al., 1984); therefore, the biological significance of the 12-month findings is uncertain. Other testicular effects were inconsistent across rat studies. Based on the evidence of testicular degeneration in male mice reported by Lish et al. (1984), there is suggestive evidence of male reproductive effects associated with RDX exposure.

Developmental studies in rats (Angerhofer et al., 1986; Cholakis et al., 1980) demonstrated effects on offspring survival, growth, and morphological development only at doses associated with severe maternal toxicity and mortality. No dose-related developmental effects were observed in rabbits (Cholakis et al., 1980). As noted in EPA's *Guidelines for Developmental Toxicity Risk Assessment* (U.S. EPA, 1991), where adverse developmental effects are produced only at doses that cause minimal maternal toxicity, developmental effects should not be discounted as being secondary to maternal toxicity; however, at doses causing excessive toxicity, as is the case with RDX, information on developmental effects may be difficult to interpret and of limited value. Therefore, at this time, no conclusions are drawn regarding developmental effects as a human hazard of RDX exposure.

1.2.4. Liver Effects

One occupational epidemiology study examined the association between RDX exposure and changes in serum liver enzymes. Case reports involving accidental exposure to RDX provide information on the potential for acute exposure to RDX to affect the liver in humans. In addition, organ weight, histopathology, and serum chemistry findings from experimental animal studies involving subchronic and chronic exposure to ingested RDX provide data relevant to an examination of the association between RDX exposure and liver effects. A summary of the liver effects associated with RDX exposure is presented in Tables 1-11 and 1-12 and Figure 1-5. Experimental animal studies are ordered in the evidence table and exposure-response array by duration of exposure and then by species.

Reports in humans provide inconsistent evidence of liver toxicity associated with acute exposure to RDX. Elevated serum levels of aspartate aminotransferase (AST) and/or alanine aminotransferase (ALT) were reported in several case reports of individuals who ingested unknown amounts of RDX (Küçükardali et al., 2003; Woody et al., 1986; Knepshield and Stone, 1972; Hollander and Colbach, 1969; Stone et al., 1969; Merrill, 1968) (see Appendix C, Section C.2). Liver biopsies did not reveal any abnormal observations (Stone et al., 1969). In other case reports, no significant changes in serum levels of liver enzymes were observed (Testud et al., 1996a; Ketel and Hughes, 1972). In a cross-sectional epidemiologic study of workers from five U.S. Army munitions plants (69 exposed to RDX alone and 24 to RDX and HMX; RDX exposure range:

undetectable (<0.01 mg/m^3) to 1.6 mg/m^3) (Hathaway and Buck, 1977), serum chemistry analysis (including the serum liver enzymes AST, ALT, and alkaline phosphatase [ALP]) revealed no statistically significant differences between exposed and unexposed workers (Table 1-11).

In experimental animals, some, but not all, subchronic studies reported increased liver weight associated with RDX exposure (Table 1-12 and Figure 1-5). Dose-related increases in relative liver weight[10] (11–25% in high-dose groups) were observed in male and female B6C3F$_1$ mice given RDX in the diet for 90 days (Cholakis et al., 1980) and in female F344 rats in two separate 90-day dietary studies of RDX (Levine et al., 1990; Levine et al., 1981a, b; Cholakis et al., 1980); however, relative liver weights were not increased in female F344 rats in another 90-day gavage study (Crouse et al., 2006). Male F344 rats exhibited an increase in relative liver weight only in one of these subchronic studies (Levine et al., 1990; Levine et al., 1981a, b). In subchronic studies in other species, absolute liver weights were increased in male and female monkeys (6–16% relative to control at 1 and 10 mg/kg-day) (Martin and Hart, 1974) and in male, but not female, beagle dogs (53% relative to control in male dogs at 10 mg/kg-day) (Hart, 1974).

Chronic RDX exposures in B6C3F$_1$ mice and F344 or Sprague-Dawley rats showed a less consistent pattern of liver weight increases. Interpretation of liver weight increases in the 2-year mouse study is complicated by the incidence of adenomas and carcinomas in each dose group; the apparent increase in liver weights in male and female mice exposed to RDX in diet (Lish et al., 1984) was reduced when mice with liver adenomas or carcinomas were removed from the analysis. In a 2-year rat study (Levine et al., 1983b), relative liver weights were increased in high-dose (40 mg/kg-day) males and females (by 11 and 18% compared to controls, respectively), likely reflecting the depressed weight gain in the high-dose rats (2 - 30% in males and 10 - 15% in females). In evaluating organ weight data across studies of all durations, less weight is placed on evidence of organ weight changes from chronic (2-year) studies because normal physiological changes associated with aging and intercurrent disease contributes to inter-animal variability that could confound organ weight interpretation (Sellers et al., 2007), as is true of the mouse liver weight data for RDX.

Nonneoplastic histopathological changes in the liver were not associated with RDX exposure in the majority of experimental animal studies (Crouse et al., 2006; Levine et al., 1990; Lish et al., 1984; Levine et al., 1983b; Levine et al., 1981a, b; Hart, 1976, 1974; Martin and Hart, 1974), including 2-year oral studies in mice at doses up to 100 mg/kg-day (Lish et al., 1984) and in rats at doses up to 40 mg/kg-day (Levine et al., 1983b). The few findings of liver lesions were

[10]Based on an evaluation of the relationship between organ weight and body/brain weight to determine which endpoint (organ weight, organ-to-body weight ratio, or organ-to-brain weight ratio) is likely to more accurately detect target organ toxicity, Bailey et al. (2004) concluded that evaluation of the effects of a test chemical on liver weight are optimally analyzed using organ-to-body weight ratios. Therefore, the analysis of liver weight here focuses on relative weight data where study authors reported both relative and absolute weights, although both relative and absolute data are summarized in the evidence table (Table 1-12).

reported in studies with more limited histopathological analyses, and were not confirmed in the studies with more complete histopathologic examination and longer exposure durations (Lish et al., 1984; Levine et al., 1983b). For example, the incidence of liver portal inflammation was increased in female rats, but not male rats, exposed to 40 mg/kg-day in the diet for 90 days (Cholakis et al., 1980). There was an increase in the incidence of mild liver microgranulomas in female mice only (Cholakis et al., 1980) and karyomegaly of hepatocytes in male mice only exposed to 320 mg/kg-day RDX in the diet for 90 days (Cholakis et al., 1980). Because both the rat and mouse studies by Cholakis et al. (1980) used relatively small group sizes (n = 10/sex/group) and provided histopathologic findings for the control and high-dose groups only, less weight is placed on these findings than on those from the 2-year bioassays. It should be noted that exposure to HMX, the primary contaminant in several of the RDX studies, was associated with histopathological changes in the livers of male rats fed doses ≥450 mg/kg-day for 13 weeks. However, similar findings were not observed in the RDX studies, where the doses of RDX employed in the studies would have resulted in HMX exposures of ≤60 mg/kg-day. The contribution of HMX exposure to the overall liver findings in the studies of RDX toxicity is therefore expected to be negligible.

Clinical chemistry parameters, including serum ALT, AST, and ALP, showed no treatment-related changes indicative of liver toxicity. Statistically significant changes in these parameters in some subchronic and chronic toxicity studies in rats and mice were relatively small (generally <50% of the control mean), were not dose-related in most instances, and showed no consistent pattern of change between sexes or across studies.

Some subchronic and chronic oral toxicity studies in rats and mice reported dose-related changes in serum cholesterol and triglyceride levels; however, these changes were not consistently observed in males and females within the same study, and patterns of changes were not consistent across studies. Specifically, serum triglyceride levels were elevated (up to 41%) in female B6C3F_1 mice exposed to RDX in the diet for 2 years, although increases were not dose-related (Lish et al., 1984); male mice in the same study did not show a similar increase in triglycerides. In contrast, serum triglycerides showed dose-related decreases in male and female F344 rats (50–62% at the high doses) in a subchronic oral (dietary) study (Levine et al., 1990; Levine et al., 1981a, b). In a chronic toxicity study by the same investigators (Levine et al., 1983b), serum triglyceride levels were generally decreased in male and female rats (52 and 51%, respectively, at the highest dose of 40 mg/kg-day); however, triglyceride levels across the four dose groups in this study did not show a dose-related response.

Serum cholesterol levels showed a dose-related increase (38% at the high dose of 100 mg/kg-day) in female B6C3F_1 mice exposed to RDX in the diet for 2 years (Lish et al., 1984); however, changes in cholesterol in male mice in the same study were not dose related. Changes in serum cholesterol in male and female F344 rats exposed to RDX in the diet for 2 years at doses up to 40 mg/kg-day (Levine et al., 1983b), in rats exposed to RDX by gavage for 90 days at doses up to

15 mg/kg-day (Crouse et al., 2006), and in monkeys exposed to RDX in the diet for 90 days (Martin and Hart, 1974) were relatively small (within 38% of control mean) and were not dose related.

Table 1-11. Evidence pertaining to liver effects in humans

Reference and study design	Results			
Hathaway and Buck (1977) (United States) Cross-sectional study, 2,022 workers, 1,491 participated (74% response rate). Analysis group: limited to whites; 69 exposed to RDX alone and 24 exposed to RDX and HMX; 338 not exposed to RDX, HMX, or TNT. **Exposure measures**: Exposure determination based on job title and industrial hygiene evaluation. Exposed subjects assigned to two groups: <LOD or ≥0.01 mg/m^3 (mean for employees with exposures ≥LOD: 0.28 mg/m^3). **Effect measures**: Liver function tests. **Analysis**: Types of statistical tests were not reported (assumed to be t-tests for comparison of means and χ^2 tests for comparison of proportions).	**Mean laboratory values of liver enzymes in men** *(mean; standard deviation not reported)*			
	Test	Referent (n = 237)	RDX exposed‡ Undetected (<LOD) (n = 22)	RDX exposed‡ >0.01 mg/m^3 (n = 45)
	LDH	173	191	174
	ALP	82	78	80
	ALA (SGOT)	22	25	21
	AST (SGPT)	21	26	18
	Bilirubin	0.5	0.4	0.4
	‡Includes both workers exposed to RDX alone and RDX and HMX. No differences were statistically significant as reported by study authors. Similar results in women.			
	Liver function tests in men *(prevalence of abnormally elevated values)*			
	Test (abnormal range)	Referent	RDX exposed‡ Undetected (<LOD)	RDX exposed‡ >0.01 mg/m^3
	LDH (>250)	2/237	1/22	0/45
	ALP (>1.5)	34/237	1/22	6/45
	AST (SGOT) (>35)	20/237	4/22	2/45
	ALT (SGPT) (>35)	15/237	2/22	0/45
	Bilirubin (>1.0)	5/237	1/22	1/45
	‡Includes both workers exposed to RDX alone and RDX and HMX. No differences were statistically significant as reported by study authors. Similar results in women.			

LDH = lactate dehydrogenase; SGOT = glutamic oxaloacetic transaminase; SGPT = glutamic pyruvic transaminase

Table 1-12. Evidence pertaining to liver effects in animals

Reference and study design	Results					
Liver weight						
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100
	Absolute liver weight at 104 wks *(percent change compared to control)*					
	M	0%	28%*	11%	12%	35%*
	F	0%	7%	7%	15%	18%*
	Relative liver weight at 104 wks *(percent change compared to control)*					
	M	0%	32%*	12%	14%	46%*
	F	0%	6%	8%	18%	45%*
	Note: Percent change in liver weights of male and female mice was reduced in all dose groups when mice with liver tumors were removed from the analysis, suggesting no real effect on liver weight.					
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Doses	0	1.0	3.1	10	
	Absolute liver weight *(percent change compared to control)*					
	M	0%	−6%	−6%	−6%	
	F	0%	7%	−11%	1%	
	Relative liver weight *(percent change compared to control)*					
	M	0%	−5%	−2%	−3%	
	F	0%	17%	−2%	13%	
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Absolute liver weight at 105 wks *(percent change compared to control)*					
	M	0%	3%	−7%	1%	−8%
	F	0%	1%	−4%	3%	0%
	Relative liver weight at 105 wks *(percent change compared to control)*					
	M	0%	1%	0%	2%	11%
	F	0%	1%	−2%	6%	18%*
Cholakis et al. (1980) Mice, B6C3F$_1$, 10–12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size Experiment 1: 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute liver weight *(percent change compared to control)*						
	M	0%	–	–	–	−6%	−5%
	F	0%	–	–	–	−4%	−1%
	Relative liver weight *(percent change compared to control)*						
	M	0%	–	–	–	−4%	−4%
	F	0%	–	–	–	−6%	1%

Reference and study design	Results						
Experiment 2: 0, 40, 60, or 80 mg/kg-d for 2 wks followed by 0, 320, 160, or 80 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Doses	0		80	160		320
	Absolute liver weight *(percent change compared to control)*						
	M	0%		2%	12%		26%*
	F	0%		4%	9%		29%*
	Relative liver weight *(percent change compared to control)*						
	M	0%		0%	9%		25%*
	F	0%		4%	4%		22%*
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Absolute liver weight *(percent change compared to control)*						
	M	0%	–	–	–	−2%	−5%
	F	0%	–	–	–	6%	4%
	Relative liver weight *(percent change compared to control)*						
	M	0%	–	–	–	2%	3%
	F	0%	–	–	–	10%	11%
Cholakis et al. (1980) Rats, CD, two-generation study; F0: 22/sex/group; F1: 26/sex/group; F2: 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size F0 and F1 parental animals: 0, 5, 16, or 50 mg/kg-d Diet F0 exposure: 13 wks pre-mating, and during mating, gestation, and lactation of F1; F1 exposure: 13 wks after weaning, and during mating, gestation, and lactation of F2; F2 exposure: until weaning	Doses	0		5	16		50
	Absolute liver weight *(percent change compared to control)*						
	M	0%		7%	−16%		–
	F	0%		0%	−14%		–
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Absolute liver weight *(percent change compared to control)*						
	M	0%	−6%	−9%	0%	7%	5%
	F	0%	1%	7%	18%*	15%	28%*
	Relative liver weight *(percent change compared to control)*						
	M	0%	0%	−1%	2%	5%	2%
	F	0%	1%	−2%	2%	−3%	2%

Reference and study design	Results						
(Levine et al. (1990); Levine et al. (1981a), 1981b))[b] Rats, F344, 3–4 wks old; 10/sex/group; 30/sex/group for controls 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Doses	0	10	30	100	300	600
	Absolute liver weight *(percent change compared to control)*						
	M	0%	5%	−1%	−2%	–	–
	F	0%	2%	4%	16%*	–	–
	Relative liver weight *(percent change compared to control)*						
	M	0%	9%	6%	20%	–	–
	F	0%	3%	5%	19%*	–	–
Hart (1974)[c] Dogs, Beagle, 3/sex/group Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	Doses	0	0.1	1	10		
	Absolute liver weight *(percent change compared to control)*						
	M	0%	–	–	53%		
	F	0%	–	–	3%		
Martin and Hart (1974)[c] Monkeys, Cynomolgus or Rhesus [d], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	Doses	0	0.1	1	10		
	Absolute liver weight *(percent change compared to control)*						
	M + F	0%	2%	6%	16%		
Histopathological lesions							
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Histopathological lesions in liver other than adenomas and carcinomas were not significantly different compared to controls, as reported by study authors.						
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Histopathological examination performed only for controls and 10 mg/kg-d rats; no significant differences compared to controls were reported by study authors.						

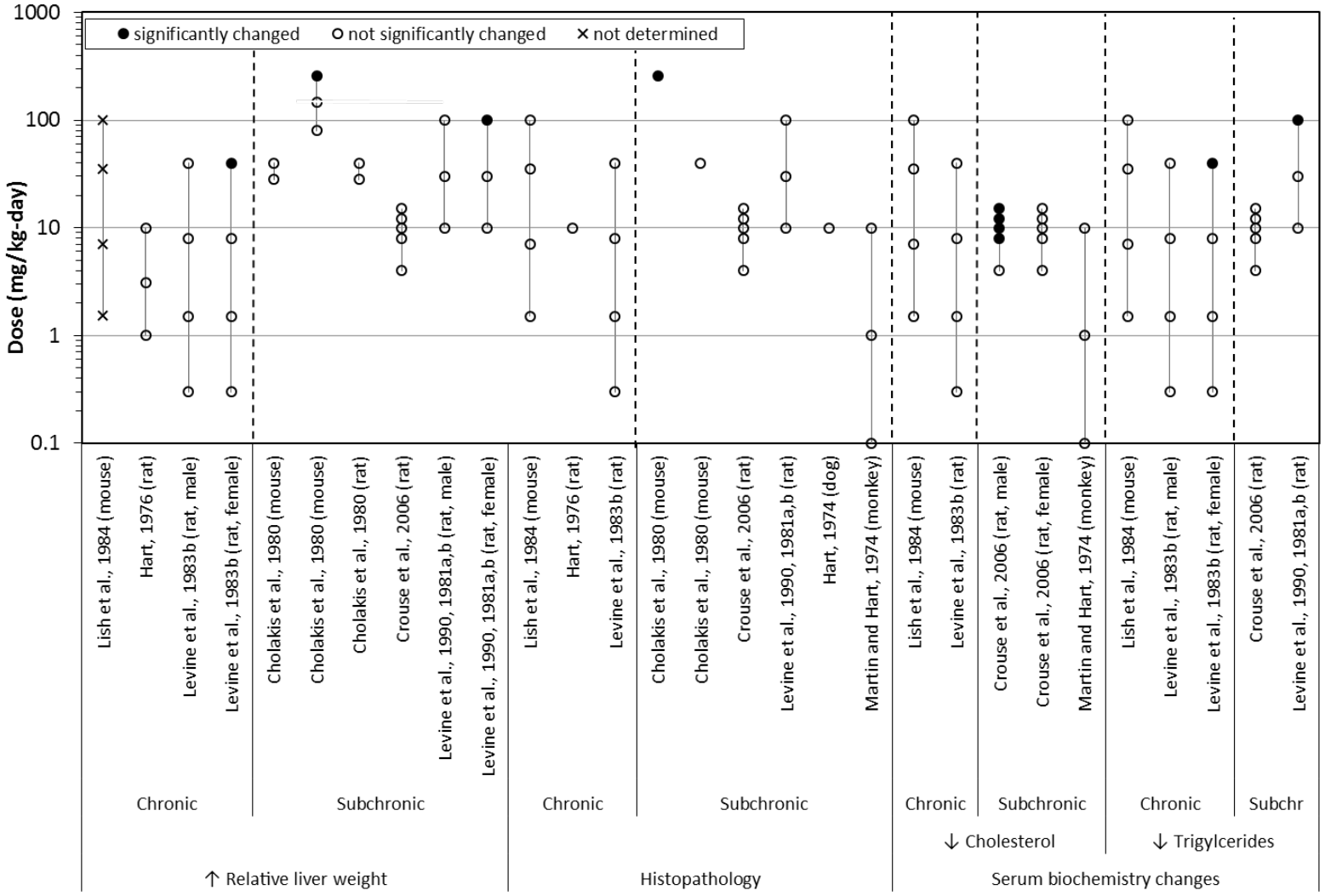
Reference and study design	Results					
Levine et al. (1983b) Rats, F344, 3–4 wks old; 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Microgranulomas *(incidence)*					
	M	0/38	0/36	0/25	0/29	0/4
	F	10/43	19/45	12/42	17/41	4/28
Cholakis et al. (1980) Mice, B6C3F$_1$, 10–12/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 80, 60, or 40 mg/kg-d for 2 wks followed by 0, 80, 160, or 320 mg/kg-d (TWA doses of 0, 79.6, 147.8, or 256.7 mg/kg-d for males and 0, 82.4, 136.3, or 276.4 mg/kg-d for females)[a] Diet 13 wks	Doses	0	80	160	320	
	Liver microgranulomas; mild *(incidence)*					
	M	2/10	–	–	1/9	
	F	2/11	–	–	7/11*	
	Increased karyomegaly of hepatocytes *(incidence)*					
	M	0/10	–	–	5/9*	
	F	–	–	–	–	
Cholakis et al. (1980) Rats, F344, 10/sex/group 88.6% pure, with 9% HMX and 2.2% water as contaminants; ~200 µm particle size 0, 10, 14, 20, 28, or 40 mg/kg-d Diet 13 wks	Doses	0	10	14	20	28	40
	Liver granulomas; mild *(incidence)*						
	M	0/10	–	–	–	–	1/10
	F	–	–	–	–	–	–
	Liver portal inflammation *(incidence)*						
	M	2/10	–	–	–	–	3/10
	F	1/10	–	–	–	–	7/10
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Histopathology examination of the 15 mg/kg-d group showed one male with mild liver congestion and one female with a moderate-sized focus of basophilic cytoplasmic alteration; neither finding was attributed by study authors to RDX treatment.						
(Levine et al. (1990); Levine et al. (1981a), 1981b))[b] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Histopathological examination of liver did not reveal any significant differences compared to controls, as reported by study authors. No histopathology findings available for the 300 or 600 mg/kg-d dose groups because all rats in these groups died before the 13-wk necropsy.						

Reference and study design	Results					
Hart (1974) Dogs, Beagle, 3/sex/group Pre-mix with ground dog chow containing 20 mg RDX/g-chow, 60 g dog food; purity and particle size not specified 0, 0.1, 1, or 10 mg/kg-d Diet 13 wks	Histopathological examination performed only for controls and 10 mg/kg-d dogs; no significant differences compared to controls were reported.					
Martin and Hart (1974) Monkeys, Cynomolgus or Rhesus[d], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	An increase in the amount of iron-positive material in liver cord cytoplasm was reported in monkeys treated with 10 mg/kg-d RDX, which the study authors considered to be of uncertain toxicological significance. Because iron-positive stain was present in controls and no further characterization of the staining was provided in the study report, the toxicological significance of this finding could not be determined.					
Serum chemistry						
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100
	Serum cholesterol at 105 wks *(percent change compared to control)*					
	M	0%	11%	−11%	5%	39%
	F	0%	5%	15%	25%	38%
	Serum triglycerides at 105 wks *(percent change compared to control)*					
	M	0%	21%	−20%	10%	−25%
	F	0%	34%	28%	41%	28%
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Serum cholesterol at 104 wks *(percent change compared to control)*					
	M	0%	15%	38%	19%	−6%
	F	0%	6%	3%	−7%	−9%
	Serum triglycerides at 104 wks *(percent change compared to control)*					
	M	0%	14%	−15%	−12%	−52%
	F	0%	18%	5%	−42%	−51%*
Crouse et al. (2006) Rats, F344, 10/sex/group 99.99% pure 0, 4, 8, 10, 12, or 15 mg/kg-d Gavage 13 wks	Doses	0	4	8	10	12	15
	Serum cholesterol *(percent change compared to control)*						
	M	0%	−3%	−10%*	−16%*	−18%*	−11%*
	F	0%	−1%	−8%	−4%	−4%	−1%
	Serum triglycerides *(percent change compared to control)*						
	M	0%	1%	1%	−7%	−2%	−19%

Reference and study design	Results						
	F	0%	−16%	−21%	7%	−37%	18%
(Levine et al. (1990); Levine et al. (1981a), 1981b))[b] Rats, F344, 10/sex/group; 30/sex for control 84.7 ± 4.7% purity, ~10% HMX, median particle diameter 20 µm, ~90% of particles ≤66 µm 0, 10, 30, 100, 300, or 600 mg/kg-d Diet 13 wks	Doses	0	10	30	100	300	600
	Serum triglyceride levels *(percent change compared to control)*						
	M	0%	−14%	−34%	−62%*	–	–
	F	0%	−12%	−29%	−50%*	–	–
Martin and Hart (1974) Monkeys, Cynomolgus or Rhesus[d], 3/sex/group Purity of test material not specified 0, 0.1, 1, or 10 mg/kg-d Gavage 13 wks	Serum biochemistry analysis revealed scattered deviations, but study authors indicated they appear to have no toxicological significance.						
	Doses	0		0.1	1		10
	Serum cholesterol *(percent change compared to control)*						
	M	0%		−17%	−2%		−7%
	F	0%		7%	7%		7%

*Statistically significant ($p < 0.05$) based on analysis by study authors.
[a]Doses were calculated by the study authors.
[b]Levine et al. (1981a) is a laboratory report of a 13-week study of RDX in F344 rats; two subsequently published papers (Levine et al., 1990; Levine et al., 1981b) present subsets of the data provided in the full laboratory report.
[c]Liver weight data from the Hart (1974) and Martin and Hart (1974) studies were considered less informative than other studies. Hart (1974) reported organ weight data for high-dose dogs (3/sex/group) only, and the liver weights from Martin and Hart (1974) were highly variable across monkeys (e.g., liver weights for the control animals ranged from 46 to 141 g). Therefore, liver weight data from these two studies were not presented in the exposure-response array for liver effects (Figure 1-5).
[d]The species of monkey used in this study was inconsistently reported in the study as either Cynomolgus (in the methods section) or Rhesus (in the summary).

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.



Note: Filled circle indicates that response was statistically significantly different from the control.
X - Not considered due to confounding caused by presence of tumors.

Figure 1-5. Exposure response array of liver effects following oral exposure.

Integration of Liver Effects

There is limited evidence from human studies and from studies in experimental animals that RDX may affect the liver. The observation of short-term elevations of serum liver enzymes in several human case reports of individuals who ingested unknown amounts of RDX suggests that RDX might target the liver; however, serum liver enzymes were not elevated in a small cross-sectional study of munition plant workers exposed to RDX. In experimental animals, dose-related increases in liver weight were observed in some studies following subchronic oral exposure, but liver weight changes were not consistent across sexes within a study or across different studies. Changes in serum chemistry were not consistent across studies and the magnitude of change relative to concurrent controls was not indicative of liver damage. Nonneoplastic histopathologic lesions of the liver were also not consistently associated with RDX exposure. At this time, no conclusions are drawn regarding liver effects as a human hazard of RDX exposure.

1.2.5. Carcinogenicity

The relationship between exposure to RDX and cancer in human populations has not been investigated. The carcinogenicity of RDX has been examined in one oral chronic/carcinogenicity bioassay in mice (Lish et al., 1984) and two bioassays in rats (Levine et al., 1983b; Hart, 1976). The 2-year studies by Lish et al. (1984) and Levine et al. (1983b) included comprehensive histopathological examination of major organs, multiple dose groups and a control, and >50 animals/dose group (plus additional interim sacrifice groups). In both studies, the maximum tolerated dose was reached or exceeded in high-dose animals (based on decreased terminal body weight in high-dose male and female mice of 5 and 19%, respectively, and decreased survival in male and female rats by approximately 50 and 25%, respectively, compared to the control). The earlier Hart (1976) study is largely limited by the lack of characterization of the test material and the pathology examination in control and high-dose groups only. A temperature spike in the animal rooms on study day 76 resulted in significant mortality across all dose groups and control animals; however, there were still >80 rats/sex/group after the overheating incident and ≥50 rats/sex/group at termination, and it seems unlikely that the mortality associated with the temperature spike would have affected a tumor response in the rats. A summary of the evidence for liver and lung tumors in experimental animals from these three bioassays is provided in Tables 1-13 and 1-14.

Liver Tumors

An increased incidence of liver tumors was observed in one chronic mouse study (Lish et al., 1984) and one of two chronic rat studies (Levine et al., 1983b). Incidences of hepatocellular tumors are presented in Table 1-13 and discussed in further detail below.

The incidence of hepatocellular carcinomas and the combined incidence of hepatocellular adenomas or carcinomas showed a statistically significant positive trend with RDX dose in female, but not male, B6C3F$_1$ mice as compared to concurrent controls in a 2-year dietary study (Lish et al.,

1984). In female B6C3F$_1$ mice, Lish et al. (1984) observed that the liver tumor incidence in the concurrent female control mice was relatively low (1/65), and significantly lower than the incidence from historical controls (historical incidence data not provided by study authors). The study authors also compared liver tumor incidence in RDX-exposed female mice to mean historical control incidence for female mice of the same strain from National Toxicology Program (NTP) studies conducted during the same time period (147/1,781 or 8%; range: 0–20%) (Haseman et al., 1985).[11] The combined incidence of hepatocellular adenomas or carcinomas in female mice at RDX doses ≥35 mg/kg-day (19% at both doses) was statistically significantly elevated when statistical analysis was performed using NTP historical control data; limitations associated with comparisons to historical control data originating from a different laboratory are acknowledged given cross-study differences in diet, laboratory, pathological evaluation, and animal provider.

A Pathology Working Group (PWG) with substantial participation by NTP pathologists reviewed the slides of female mouse liver lesions from the Lish et al. (1984) study (Parker et al., 2006; Parker, 2001). Some malignant tumors were downgraded to benign status and several lesions initially characterized as adenomas were changed to non-neoplastic lesions based on more recent diagnostic criteria used by the PWG (Harada et al., 1999). There remained a statistically significant positive trend in the combined incidence of hepatocellular adenomas or carcinomas, consistent with the original findings of Lish et al. (1984). Because the PWG analysis reflects more recent histopathological criteria for the grading of tumors, the incidence of hepatocellular adenomas or carcinomas as reported by Parker et al. (2006) were considered the more reliable measure of liver tumor response in female mice from the Lish et al. (1984) bioassay.

In male mice from the Lish et al. (1984) study, the incidences of hepatocellular carcinomas in treated groups were higher than in the control, and the combined incidences of hepatocellular adenomas or carcinomas of male mice were higher in three of four treated groups than in the control; however, there were no statistically significant trends in either case. The incidences of liver carcinoma in control (21%) and treated groups of male mice (22–33%) were generally within the range for the same mouse strain reported by NTP (8–32%) (Haseman et al., 1985). Similarly, the combined incidences of liver adenoma or carcinoma in control (32%) and treated groups

[11]Comparison of control incidences of hepatocellular adenomas or carcinomas between Lish et al. (1984) and Haseman et al. (1985) must be interpreted with caution because of cross-study differences in labs, diets, and sources of animals. Specifically, the labs used by NTP and analyzed by Haseman et al. (1985) did not include the lab contracted to perform the Lish et al. (1984) study, and it is not clear if the diet used in the Lish et al. (1984) study was included in the diets reported in the NTP studies. Further, the NTP studies included three different suppliers of mice; one supplier was also used in the Lish et al. (1984) study. EPA *Guidelines for Carcinogenic Risk Assessment* (U.S. EPA, 2005a) also note that, unless the tumor is rare, the standard for determining statistical significance of tumor incidence is a comparison of dosed animals with the concurrent controls.

(27–48%) were within the range for the same mouse strain reported by NTP (14–58%) (Haseman et al., 1985).[12] The PWG did not re-analyze liver tumor slides from male mice.

A statistically significant positive trend with dose was observed in the incidence of hepatocellular carcinomas in male, but not female, F344 rats exposed to RDX in the diet for 2 years (Levine et al., 1983b). In the Levine et al. (1983b) study, there were only a few tumors observed in the exposed groups of male rats (0/55, 0/52, 2/55, 2/31) relative to the control (1/55), and inferences made from such a sparse response are uncertain. Because liver tumors are rare tumors in the rat[13], some perspective is obtained by considering historical control data. In a paper published concurrently with the Levine et al. (1983b) study, NTP reported an incidence of liver carcinomas in untreated control male F344 rats of 0.7% (12/1,719; range: 0–2%) (Haseman et al., 1985). In Levine et al. (1983b), the incidence of liver carcinomas in control male rats (1/55 or 1.8%) was at the upper end of this NTP range, and the incidence in RDX-treated male F344 rats in the highest two dose groups (3.6 and 6.4%) exceeded the NTP historical control range. Using incidence data from NTP historical controls, the trend for carcinoma in the RDX-treated F344 rats was statistically significant (p-value = 0.003; one-sided exact Cochrane-Armitage trend test). It should be noted that although the NTP historical controls (Haseman et al., 1985) are comparable with Levine et al. (1983b) in terms of the time period, they may not be directly comparable in terms of diet, laboratory, pathological evaluation, and animal provider. However, other historical control datasets from male F344 rats, both recent and of the time period of the Levine study, indicate similar low incidences of liver carcinomas (0.36%, (NTP, 2009); 0.31%, (Maita et al., 1987)). In the Levine et al. (1983b) study, the mortality in the highest dose group is substantially higher than in the other dose groups during the second year leading to uncertainty in the true cancer incidence in the high dose group. It was not possible to estimate mortality-adjusted incidences because no time-to-death information was available.

Nonmalignant liver tumors (neoplastic nodules) in F344 male rats in NTP historical controls were reported more frequently than carcinomas, with an average incidence of 3.5% (61/1,719; range: 0–12%) (Haseman et al., 1985); Levine et al. (1983b) reported an incidence of neoplastic nodules of 7.3% in their control male rats, consistent with the NTP historical control data, and a decline in incidence with increasing RDX exposure. The combined incidence of liver neoplastic nodules or carcinomas did not show a significant trend with dose.

In a second 2-year dietary study in the rat study using a different strain (Sprague-Dawley), the combined incidence of hepatocellular adenomas or carcinomas was not increased with dose in rats of either sex at doses up to 10 mg/kg-day (Hart, 1976). However, interpretation of results

[12]Ibid.

[13]NTP historical control data for hepatocellular carcinomas F344 rats as reported in Haseman et al. (1985): 12/1,719 (0.7%) in males; 3/1,766 (0.17%) in females. Historical control data for Charles River Sprague-Dawley rats as reported in Chandra et al. (1992): 6/1,340 (0.45%) in males; 1/1,329 (0.08%) in females.

from this study is limited by the comparatively lower doses employed in the study, and the recording of effects only at the control and high dose groups.

Table 1-13. Liver tumors observed in chronic animal bioassays

Reference and study design	Results[a]					
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100[b]
	Hepatocellular adenomas *(incidence)*[c]					
	M	8/63 (12.7)	6/60 (10.0)	1/62* (1.6)	7/59 (11.9)	7/27 (25.9)
	F	1/65 (1.5)	1/62 (1.6)	6/64 (9.4)	6/64 (9.4)	3/31 (9.7)
	Hepatocellular carcinomas *(incidence)*[c]					
	M	13/63 (20.6)	20/60 (33.3)	16/62 (25.8)	18/59 (30.5)	6/27 (22.2)
	F	0/65 (0.0)	4/62 (6.5)	3/64 (4.7)	6/64 (9.4)	3/31[d] (9.7)
	Hepatocellular adenoma or carcinoma combined *(incidence)*[c]					
	M	20/63 (31.7)	26/60 (43.3)	17/62 (27.4)	25/59 (42.4)	13/27 (48.1)
	F	1/65 (1.5)	5/62 (8.1)	9/64* (14.1)	12/64* (18.8)	6/31*[d] (19.4)
	PWG reanalysis of liver lesion slides from female mice (Parker et al., 2006; Parker, 2001).[e]					
	Doses	0	1.5	7.0	35	175/100[b]
	Hepatocellular adenomas *(incidence)*[c]					
	F	1/67 (1.5)	3/62 (4.8)	2/63 (3.2)	8/64 (12.5)	2/31 (6.5)
	Hepatocellular carcinomas *(incidence)*[c]					
	F	0/67 (0.0)	1/62 (1.6)	3/63 (4.8)	2/64 (3.1)	2/31 (6.5)
	Hepatocellular adenoma or carcinoma combined *(incidence)*[c]					
	F	1/67 (1.5)	4/62 (6.5)	5/63 (7.9)	10/64 (15.6)	4/31[d] (12.9)

Reference and study design	Results[a]				
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	**Doses**	0	1.0	3.1	10
	Neoplastic nodules *(incidence)*[c]				
	M	0/82	–	–	3/77
	F	1/72	–	–	1/81
	Hepatocellular carcinomas *(incidence)*[c]				
	M	1/82	–	–	1/77
	F	1/72	–	–	1/81[f]
	Neoplastic nodules or hepatocellular carcinomas combined *(incidence)*[c]				
	M	1/82	–	–	4/77
	F	2/72	–	–	2/81
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Neoplastic nodules *(incidence)*[c]					
	M	4/55 (7.3)	3/55 (5.5)	0/52 (0.0)	2/55 (3.6)	1/31 (3.2)
	F	3/53 (5.6)	1/55 (1.8)	1/54 (1.9)	0/55 (0.0)	4/48 (8.3)
	Hepatocellular carcinomas *(incidence)*[c]					
	M	1/55 (1.8)	0/55 (0.0)	0/52 (0.0)	2/55 (3.6)	2/31[d] (6.5)
	F	0/53 (0.0)	1/55 (1.8)	0/54 (0.0)	0/55 (0.0)	0/48 (0.0)
	Neoplastic nodules or hepatocellular carcinomas combined *(incidence)*[c]					
	M	5/55 (9.1)	3/55 (5.5)	0/52 (0.0)	4/55 (7.3)	3/31 (9.7)
	F	3/53 (5.6)	2/55 (3.6)	1/54 (1.9)	0/55 (0.0)	4/48 (8.3)

*Statistically significant difference compared to the control group ($p < 0.05$), identified by the authors.
[a]Selected percent incidences are provided in parentheses below the incidences to help illustrate patterns in the responses.
[b]The lower dose of 100 mg/kg-day was started in week 11, resulting in a duration-weighted average dose of 107 mg/kg-day.
[c]The incidences reflect the animals surviving to month 12.
[d]Statistically significant trend ($p < 0.05$) was identified using a one-sided Cochran-Armitage trend tests performed by EPA.
[e]The numbers of animals at risk (i.e., the denominators) in the control group (n = 67) and 7 mg/kg-day dose group (n = 63) as reported in the PWG reanalysis (Parker et al., 2006; Parker, 2001) differed from the numbers reported in the original study by Lish et al. (1984) (n = 65 and 64, respectively). Further investigation of these differences by the U.S. Army (sponsor of the mouse bioassay and subsequent PWG reevaluation) was unable to resolve the

discrepancy (email to Louis D'Amico, U.S. EPA, from Mark Johnson, U.S. Army Public Health Command, February 13, 2015).

[f]Hart (1976) distinguishes the single high-dose carcinoma in the liver from a hepatocellular carcinoma; the incidence of hepatocellular carcinomas in this dose group is shown as 0/81 (p. 119 of the publication).

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.

Lung Tumors

Lung tumors were observed in female and male B6C3F$_1$ mice exposed to RDX in the diet for 2 years (Lish et al., 1984) (see Table 1-14). Incidence of alveolar/bronchiolar carcinomas and the combined incidence of alveolar/bronchiolar adenomas or carcinomas showed a statistically significant positive trend (one-sided *p*-values of 0.016 and 0.009, respectively, for the Cochran-Armitage trend test) in female mice. Incidence of alveolar/bronchiolar carcinomas in male mice showed a statistically significant positive trend (*p*-value = 0.015; one-sided Cochran-Armitage trend test). However, the combined incidence of adenomas and carcinomas was not elevated in male mice. In such a case, NTP policy recommends analyzing the tumors both separately and in combination (McConnell et al., 1986). This recommendation arose out of concern that combining benign and malignant neoplasms can result in a false negative if the chemical shows a statistically significant increase in malignant tumors without an increase in the combined incidence. In an addendum to the study report that included results of additional examination and sectioning of lung specimens from the mid-dose groups in the mouse study, Lish et al. (1984) noted an increase in the combined incidences of primary pulmonary neoplasms in males of all dose groups and in females in the 7.0, 35, and 175/100 mg/kg-day dose groups, but regarded these neoplasms as random and not biologically significant (rationale for this conclusion not provided).

Bioassays in rats provide no evidence of an association between RDX exposure and induction of lung tumors. The incidence of alveolar/bronchiolar adenomas or carcinomas was not increased in either sex of Sprague-Dawley rats exposed chronically to RDX at doses up to 10 mg/kg-day (Hart, 1976) or in F344 rats of either sex exposed chronically to RDX at doses up to 40 mg/kg-day (Levine et al., 1983b). Alveolar/bronchiolar carcinomas are rare tumors in both species of rats, male or female (Chandra et al., 1992; Haseman et al., 1985).

Table 1-14. Lung tumors observed in chronic animal bioassays

Reference and study design	Results[a]					
Lish et al. (1984) Mice, B6C3F$_1$, 85/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 1.5, 7.0, 35, or 175/100 mg/kg-d (high dose reduced to 100 mg/kg-d in wk 11 due to excessive mortality) Diet 2 yrs	Doses	0	1.5	7.0	35	175/100[b]
	Alveolar/bronchiolar adenomas *(incidence)*[c]					
	M	6/63 (9.5)	5/60 (8.3)	5/62 (8.1)	7/59 (11.9)	1/27 (3.7)
	F	4/65 (6.2)	2/62 (3.2)	5/64 (7.8)	9/64 (14.1)	3/31 (9.7)
	Alveolar/bronchiolar carcinomas *(incidence)*[c]					
	M	3/63 (4.8)	6/60 (10.0)	3/62 (4.8)	7/59 (11.9)	5/27[d] (18.5)
	F	3/65 (4.6)	1/62 (1.6)	3/64 (4.7)	3/64 (4.7)	4/31[d] (12.9)
	Alveolar/bronchiolar adenoma or carcinoma combined *(incidence)*[c]					
	M	9/63 (14.3)	11/60 (18.3)	8/62 (12.9)	14/59 (23.7)	6/27 (22.2)
	F	7/65 (10.8)	3/62 (4.8)	8/64 (12.5)	12/64 (18.8)	7/31[d] (22.6)
Hart (1976) Rats, Sprague-Dawley, 100/sex/group Purity and particle size not specified 0, 1.0, 3.1, or 10 mg/kg-d Diet 2 yrs	Doses	0	1.0	3.1		10
	Alveolar/bronchiolar adenoma *(incidence)*					
	M	2/83	–	–		1/77
	F	0/73	–	–		0/82
	No alveolar/bronciolar carcinomas reported by study authors.					
Levine et al. (1983b) Rats, F344, 75/sex/group; interim sacrifices (10/sex/group) at 6 and 12 mo 89.2–98.7% pure, with 3–10% HMX as contaminant; 83–89% of particles <66 µm 0, 0.3, 1.5, 8.0, or 40 mg/kg-d Diet 2 yrs	Doses	0	0.3	1.5	8.0	40
	Alveolar/bronchiolar adenomas *(incidence)*[c]					
	M	1/55	0/15	1/17	0/16	1/31
	F	3/53	0/7	0/8	1/10	0/48
	Alveolar/bronchiolar carcinomas *(incidence)*[c]					
	M	–	–	–	–	–
	F	0/53	0/7	1/8	0/10	0/48
	Alveolar/bronchiolar adenoma or carcinoma combined *(incidence)*[c]					
	M	–	–	–	–	–
	F	3/53	0/7	1/8	1/10	0/48

[a]Selected percent incidences are provided in parentheses below the incidences to help illustrate patterns in the responses.
[b]The lower dose of 100 mg/kg-day was started in week 11, resulting in a duration-weighted average dose of 107 mg/kg-day.
[c]The incidences reflect the animals surviving to month 12.

[d]Statistically significant trend ($p < 0.05$) was identified using a one-sided Cochran-Armitage trend test performed by EPA.

Note: A dash ("–") indicates that the study authors did not measure or report a value for that dose group.

Mechanistic Evidence

There are few mechanistic data to inform a MOA determination for either liver or lung tumors induced by exposure to RDX.

The available in vitro and in vivo genotoxicity assay results are largely negative for parent RDX or its oxidative metabolites (see Appendix C, Section C.3.2), supporting the hypothesis that parent RDX or its oxidative metabolites do not interact directly with deoxyribonucleic acid (DNA). In contrast, there are some positive genotoxicity results for the N-nitroso metabolites of RDX, specifically hexahydro-1-nitroso-3,5-dinitro-1,3,5-triazine (MNX) and hexahydro-1,3,5-trinitroso-1,3,5-triazine (TNX). MNX and TNX have been identified from minipigs; minipigs were chosen as the animal model for investigation of RDX metabolism because the GI tract of pigs more closely resembles that of humans (Musick et al., 2010; Major et al., 2007). MNX has tested positive in some in vitro assays, including unscheduled DNA synthesis in primary rat hepatocytes and the mouse lymphoma forward mutation assay (Snodgrass, 1984), although MNX tested negative in the only in vivo test performed, a mouse dominant lethal mutation test (Snodgrass, 1984). MNX was not mutagenic in *Salmonella typhimurium* (strains TA98, TA100, TA1535, TA1537, and TA1538), with or without the addition of the S9 metabolic activating mixture (Pan et al., 2007b; Snodgrass, 1984). When *S. typhimurium* strains TA97a and TA102, strains sensitive to frame shift and oxidative DNA damage, were used in conjunction with elevated concentrations of the metabolizing system (S9), MNX and TNX were mutagenic. N-nitroso metabolites, including MNX and TNX, are generated anaerobically and are likely a result of bacterial transformation of parent RDX in the GI tract to various N-nitroso derivatives (Pan et al., 2007b). Exposure to potentially mutagenic N-nitroso metabolites of RDX generated in the GI tract of mice may occur in the liver (and subsequently in the systemic circulation) via enterohepatic circulation. However, in pigs, the N-nitroso metabolites of RDX have been identified only in trace amounts in urine compared to the major metabolites, 4-nitro-2,4-diazabutanal and 4-nitro-2,4-diaza-butanamide (Major et al., 2007). Thus, the contribution of the N-nitroso metabolites to the overall carcinogenic potential of RDX is unclear.

Aberrant expression of miRNAs was observed in the brains and livers of female $B6C3F_1$ mice fed 5 mg RDX/kg in the diet for 28 days (Zhang and Pan, 2009b) (dose of 0.75–1.5 mg/kg-day estimated by Bannon et al. (2009b)), with several oncogenic miRNAs being upregulated, while several tumor-suppressing miRNAs were downregulated. However, the pattern of induction was not always consistent in the livers of RDX-treated mice (e.g., miR-92a was downregulated in liver tissue samples when it is typically upregulated in hepatocellular carcinomas) (Sweeney et al., 2012b). miRNAs have been associated with several cancers (Wiemer, 2007; Zhang et al., 2007), but the utility of miRNAs as predictive of carcinogenesis has not been demonstrated (Bannon et al., 2009b). Further, it is unknown whether or not aberrant expression of a specific miRNA (or suite of

miRNAs) plays a role in the MOA of RDX carcinogenicity. Microarray analysis of gene expression in male Sprague-Dawley rats after exposure to a single oral (capsule) dose of RDX revealed a general upregulation in gene expression (predominantly genes involved in metabolism) in liver tissues (Bannon et al., 2009a); however, the relevance of this finding to the carcinogenicity of RDX is unclear.

Sweeney et al. (2012b) hypothesized a set of MOAs for the liver tumors:

- *Genotoxicity mediated by either: (1) RDX; (2) tissue-generated oxidative metabolites; or (3) N-nitroso metabolites generated anaerobically in the GI tract.* The key events in this hypothesized MOA are: production of DNA damage, gene mutation, formation of neoplastic lesions, and promotion/progression of tumors. The largely negative results for genotoxicity led Sweeney et al. (2012b) to conclude that this MOA is not plausible for RDX or its oxidative metabolites. Although there are some positive results for the N-nitroso metabolites, the limited evidence to support systemic uptake and distribution of metabolites to the liver led Sweeney et al. (2012b) to conclude that this MOA is not sufficiently plausible.

- *Cell proliferation.* The key events in this hypothesized MOA are GI-tract generation of N-nitroso metabolites, absorption, distribution to the liver, cytoxocity (optional), and enhanced cell proliferation, leading to preneoplastic foci that progress to hepatocellular adenomas and carcinomas. Sweeney et al. (2012b) cited evidence of increased liver weights in mice as consistent with cell proliferation, but noted that increased liver weights were also observed in rats without proceeding to liver tumors. They considered this MOA "plausible, but not particularly well supported."

In addition, there are other results that do not support a metabolite-based proliferative response as the MOA for carcinogenesis.

- The absence of significant liver histopathology in mice after subchronic or chronic exposure to RDX at doses that induced liver tumors (Lish et al., 1984; Cholakis et al., 1980) suggests that cellular toxicity is not a precursor to these tumors.

- As discussed in Section 1.2.4, changes in liver weight showed no consistent pattern across studies or sexes, and did not correlate with tumor response.

- No studies were available that directly measured RDX-induced cell proliferation rates.

- No information was available to rule out non-precancerous causes of liver weight increase.

In summary, the available evidence indicates that RDX is likely not mutagenic (see Appendix C, Section C.3.2), although anaerobically-derived N-nitroso metabolites have demonstrated some genotoxic potential. While these metabolites have been measured in the mouse (Pan et al., 2007b) and minipig (Musick et al., 2010; Major et al., 2007), they have not been identified in humans, and may not be the predominant metabolites of RDX. A MOA involving a proliferative response generated by tissue-derived oxidative metabolites of RDX has been proposed, but is not supported by the available data. In light of limited information on precursor

events leading to the observed liver and lung tumor response in RDX-exposed rodents and lack of toxicokinetic information on RDX metabolites, neither a cell proliferative MOA nor a mutagenic N-nitroso metabolite MOA is supported. Thus, the MOA leading to the increased incidence of liver and lungs tumors is not known.

1.2.6. Other Noncancer Effects

There are isolated reports of RDX inducing systemic effects in several organs/systems, including the eyes and the musculoskeletal, cardiovascular, immune, and GI systems. However, there is less evidence for these effects compared to organ systems described earlier in Section 1.2. Generally, evidence for toxicological effects in these organ systems was limited to human case reports, lacked reproduction or were not observed in other studies of similar duration in the same species, or lacked consistent, dose-related patterns of increasing or decreasing effect. A longer discussion of the evidence for each of the other noncancer effects noted above is provided in Appendix C.3.2. At this time, no conclusions are drawn regarding the other noncancer effects as human hazards of RDX exposure.

1.3. INTEGRATION AND EVALUATION

1.3.1. Effects Other Than Cancer

The majority of evidence for the health effects of RDX comes from oral toxicity studies in animals. The three epidemiology studies that document possible inhalation exposure are limited by various study design features, including inability to distinguish exposure to TNT (associated with liver and hematological system toxicity), inability to adequately characterize exposure levels, small sample sizes, and inadequate reporting. The single animal inhalation study identified in the literature search had deficiencies (e.g., lack of a control and incomplete exposure information) that precluded its inclusion in this assessment (see literature search section).

The strongest evidence for hazards following exposure to RDX is for nervous system effects. Toxicity studies in multiple animal species involving chronic, subchronic, and gestational exposures provide consistent evidence of nervous system effects following oral exposure. Effects included dose-related increases in seizures and convulsions, as well as observations of tremors, hyperirritability, hyper-reactivity, and other behavioral changes (Crouse et al., 2006; Angerhofer et al., 1986; Levine et al., 1983b; Levine et al., 1981a, b; Cholakis et al., 1980; von Oettingen et al., 1949).

Human studies provide supporting evidence for RDX as a neurotoxicant and that the nervous system effects observed in experimental animals are plausible in, and relevant to, humans. A cross-sectional study described memory impairment and visual-spatial decrements in RDX-exposed workers (Ma and Li, 1993), although confidence in these findings is relatively low because of issues with design and reporting. Several case reports provide additional evidence of associations between exposure to RDX (via ingestion, inhalation, and possibly dermal exposure)

and seizures and convulsions (Kasuske et al., 2009; Küçükardali et al., 2003; Testud et al., 1996a; Testud et al., 1996b; Woody et al., 1986 and others, see Appendix C.2). Other nervous system effects identified in human case reports include dizziness, headache, confusion, and hyperirritability.

Additional support for an association between RDX exposure and nervous system effects comes from consistent evidence of neurotoxicity across taxa, including several species of wildlife (Quinn et al., 2013; Garcia-Reyero et al., 2011; McFarland et al., 2009; Gogal et al., 2003). The association between RDX and neurological effects is biologically plausible, with studies demonstrating a correlation between blood and brain concentrations of RDX and the time of seizure onset (Williams et al., 2011; Bannon et al., 2009a). Additionally, the affinity of RDX for the picrotoxin convulsant site of the $GABA_A$ channel suggests that the resulting disinhibition could lead to the onset of seizures (Williams et al., 2011).

Induction of convulsions and seizures appears to be more strongly correlated with dose than with duration of exposure. However, there is some mechanistic information to suggest that repeated exposure to a chemical binding to the receptor convulsant site of $GABA_A$ may promote a state of increased neuronal activity that could increase the likelihood of subsequent neurological effects (Gerkin et al., 2010). It is unclear if nervous system effects progressed in severity (e.g., from behavioral change to seizures and convulsions) with increasing dose, as many of the studies that reported more subtle neurobehavioral changes did not provide detailed dose-response information, and the majority of studies were not designed to capture this information.

The nervous system effects following oral exposure to RDX were observed in humans acutely exposed to RDX and in multiple experimental animal studies in rats, mice, monkeys, and dogs following exposures ranging from 10 days to 2 years in duration. Across the database, behavioral manifestations of seizure activity were the most consistently observed nervous system effect associated with RDX exposure. This most commonly included evidence of increased convulsions, as well as other related effects such as tremors, shaking, hyperactivity, or nervousness, which were generally observed at doses that were the same as or higher than doses that induced convulsions. Nervous system effects are a human hazard of RDX exposure and are carried forward for consideration for dose-response analysis. Convulsions, considered a severe adverse effect, were selected as a consistent and sensitive endpoint representative of nervous system effects.

Evidence for kidney and other urogenital toxicity is more limited than evidence for neurotoxicity. Histopathological changes in the urogenital system (suppurative prostatitis, medullary papillary necrosis, suppurative pyelitis, uremic mineralization, and luminal distention and cystitis of the urinary bladder) were reported in male rats exposed to RDX in the diet for 2 years (Levine et al., 1983b). Similar histopathological changes of the urogenital system were not observed in mice, and no other rat studies of similar duration that examined the prostate were available. As discussed earlier, among the lesions identified in the rat, the incidence of suppurative prostatitis is considered a marker for RDX-related urogenital effects as it is plausibly associated

with a progression of effects resulting from other kidney or bladder lesions. The plausibility of a MOA that shares a common molecular initiating event (binding to the $GABA_A$ receptor convulsant-site) with the neurotoxic effects of RDX provides some support for an association between RDX exposure and kidney and other urogenital effects. Kidney and other urogenital system effects are a potential human hazard of RDX exposure and were carried forward for consideration for dose-response analysis. Prostatitis, considered a marker for the kidney and urogenital effects, was selected as a sensitive endpoint representative of the urogenital system effects.

There is some evidence for male reproductive toxicity that comes from the finding of testicular degeneration in male $B6C3F_1$ mice chronically exposed to RDX in the diet (Lish et al., 1984) in the only mouse study conducted of that duration (24 months). The effect was noted by the study authors at both the penultimate and highest dose tested in the study. However, studies in different rat strains did not consistently observe testicular effects. Although the available data are limited, given the dose-related findings of mouse testicular degeneration, there is suggestive evidence of male reproductive effects associated with RDX exposure; these effects were carried forward for consideration for dose-response analysis. Testicular degeneration, the only endpoint observed, was selected as the endpoint representative of male reproductive effects.

Evidence for developmental toxicity and liver toxicity was more limited than that for the endpoints discussed above. In animal studies, developmental effects, including offspring survival, growth, and morphological development, were observed only at doses associated with maternal mortality (Angerhofer et al., 1986; Cholakis et al., 1980). Evidence for potential hepatic effects comes from observations of increases (generally dose-related) in liver weight in some subchronic oral animal studies (Lish et al., 1984; Levine et al., 1983b; Levine et al., 1981a, b; Cholakis et al., 1980; Hart, 1976). However, these elevations in liver weight were not consistently observed across studies nor were they accompanied by RDX-related histopathological changes in the liver or increases in serum liver enzymes. In addition, the interpretation of liver weight changes in the mouse bioassay by Lish et al. (1984) is complicated by the relatively high incidence of liver tumors in this study. At this time, no conclusions are drawn regarding developmental and liver toxicity as human hazards of RDX exposure; these effects were not considered further for dose-response analysis and derivation of reference values.

As discussed in Section 1.2, mortality is not addressed in this assessment as a hazard by itself, but rather in the context of nervous and urogenital system hazards. Histopathological changes in the urogenital system observed in male rats exposed to 40 mg/kg-day in the diet for 2 years were considered the principal cause of treatment-related morbidity and mortality (Levine et al., 1983b). However, the incidence of suppurative prostatitis, considered a sensitive marker of the urogenital effects, was increased at doses of ≥1.5 mg/kg-day. Therefore, the mortality characterized as secondary to renal effects in Levine et al. (1983b) is a less sensitive endpoint (by more than 10-fold) than the effect that is the selected as the basis dose-response analysis (i.e., suppurative prostatitis).

In a number of the animal studies reporting nervous system effects, unscheduled deaths occurred at RDX doses as low as those that induced nervous system effects (Crouse et al., 2006; Angerhofer et al., 1986; Levine et al., 1983b; Levine et al., 1981a; Cholakis et al., 1980; von Oettingen et al., 1949). In a 90-day study that recorded nervous system effects and survival more thoroughly than earlier studies, Crouse et al. (2006) reported that nearly all pre-term deaths were preceded by neurotoxic signs such as tremors and convulsions. Convulsions did not, however, necessarily lead to early mortality; of the animals observed to have convulsed in the Crouse et al. (2006) study, approximately 75% survived to the end of the 90-day study. Most of the earlier studies provide a limited understanding of the association between mortality and nervous system effects because the frequency of clinical observations was likely insufficient to observe convulsions prior to death. In humans, mortality has not been reported in case reports involving workers with symptoms of neurotoxicity exposed to RDX during manufacture or in individuals exposed acutely as a result of accidental or intentional ingestion; however, survival has not been specifically evaluated in studies of worker populations exposed chronically to RDX.

Regarding mortality, the preference, in general, is not to use a frank health effect as severe as mortality as the basis for a reference value. As noted in U.S. EPA (2002), a chemical may cause a variety of effects ranging from severe—such as death—to more subtle biochemical, physiological, or pathological changes; primary attention in assessing health risk should be given to those effects in the lower exposure range and/or the effects most biologically appropriate for a human health risk assessment. Where mortality occurs as a consequence of a chemical's effects on a specific organ/system (e.g., in the case of RDX, evidence suggests some relationship between mortality and effects on the nervous or kidney/urogenital systems), the preference would be to develop a quantitative assessment based on the initial hazard and not on death. Because unscheduled deaths were observed with some consistency across studies and, in some studies, at doses as low as those associated with convulsions, two additional analyses of mortality data are presented in Chapter 2. In the first analysis, BMDs derived using mortality data sets are compared to the BMD used to derive the RfC (Section 2.1.6). In addition, the relationship between convulsions and mortality is not clear and raises concerns for the potential underreporting of convulsions (see Section 1.2.1). An analysis, described in Section 2.1.7, addresses the possibility that the analyses of convulsions brought forward for dose-response analysis resulted in an underestimate of the toxicity for RDX.

1.3.2. Carcinogenicity

As presented in Section 1.2.5, dietary administration of RDX induced dose-related increases in the incidence of hepatocellular adenomas or carcinomas in male and female B6C3F$_1$ mice mice (Parker et al., 2006; Lish et al., 1984). In the same study, RDX also induced dose-related increases in the incidence of alveolar/bronchiolar adenomas or carcinomas in both sexes. Some of these trends in liver and lung were statistically significant. In Fischer 344 rats, dietary administration of RDX yielded a statistically significant trend in the incidence of hepatocellular carcinomas in males, but not in females (Levine et al., 1983b). A 2-year dietary study in Sprague-Dawley rats was

negative in both sexes Hart (1976), although doses in this study were somewhat lower (no carcinomas at doses up to 10 mg/kg-d in Hart (1976), versus hepatocellular carcinomas at 8 and 40 mg/kg-d in the Levine et al. (1983b) study. The human studies are not informative.

This evidence leads to consideration of two hazard descriptors under the EPA's cancer guidelines (U.S. EPA, 2005a). The descriptor *likely to be carcinogenic to humans* is appropriate when the evidence is "adequate to demonstrate carcinogenic potential to humans" but does not support the descriptor *carcinogenic to humans*. One example from the cancer guidelines is "an agent that has tested positive in animal experiments in more than one species, sex, strain, site, or exposure route, with or without evidence of carcinogenicity in humans." RDX matches the conditions of this example, having induced dose-related increases in tumors in two species (mouse and rat), in both sexes, and at two sites (liver and lung).

Alternatively, the descriptor *suggestive evidence of carcinogenic potential* is appropriate when the evidence raises "a concern for potential carcinogenic effects in humans" but is not sufficient for a stronger conclusion. The hepatocellular carcinoma result in male F344 rats is based on a small number of tumors (two each at 8 and 40 mg/kg-day, versus one in controls), and RDX did not increase the incidence of carcinomas at any other site in F344 or Sprague-Dawley rats of either sex.

As noted in the EPA's cancer guidelines (U.S. EPA, 2005a), choosing a hazard descriptor cannot be reduced to a formula, as descriptors may be applicable to a variety of potential data sets and represent points along a continuum of evidence. In the case of RDX, there are plausible scientific arguments for more than one hazard descriptor. Overall, the considerations discussed above, interpreted in light of the cancer guidelines, lead to the conclusion that there is *suggestive evidence of carcinogenic potential* for RDX. Although the evidence includes dose-related tumor increases in two species, two sexes, and two sites, the evidence of carcinogenicity outside the $B6C3F_1$ mouse is not robust, and this factor was decisive in choosing a hazard descriptor. Within the spectrum of results covered by the descriptor *suggestive evidence*, the evidence for RDX is strong. There are well-conducted studies that tested large numbers of animals at multiple dose levels, making the cancer response suitable for dose-response analysis (Section 2).

The descriptor *suggestive evidence of carcinogenic potential* applies to all routes of human exposure. Dietary administration of RDX to mice and rats induced tumors of the liver or lung, sites beyond the point of initial contact, and human case reports have demonstrated absorption and distribution of inhaled RDX into the systemic circulation. Under the cancer guidelines, this information provides sufficient basis to apply the cancer descriptor developed from oral studies to other exposure routes.

1.3.3. Susceptible Populations and Lifestages for Cancer and Noncancer Outcomes

Susceptibility refers to factors such as lifestage, genetics, sex, and health status that may predispose a group of individuals to greater response to an exposure. This greater response could be achieved either through differences in exposure to the chemical underlying RDX toxicokinetics

or differences in RDX toxicodynamics between susceptible and other populations. Little information is available on populations that may be especially vulnerable to the toxic effects of RDX.

Lifestage, and in particular childhood, susceptibility has not been observed in human or animal studies of RDX toxicity. Transfer of RDX from dam to the fetus during gestation has been reported, and the presence of RDX in the milk of dams administered 6 mg/kg-day by gavage has been documented (Hess-Ruth et al., 2007); however, reproductive and developmental toxicity studies generally did not identify effects in offspring at doses below those that also caused severe maternal toxicity (Angerhofer et al., 1986; Cholakis et al., 1980). Thus, the existing toxicity literature does not provide evidence of early lifestage susceptibility to RDX.

Limited data suggest that male laboratory animals may be more susceptible to noncancer toxicity associated with RDX exposure. In general, male animals were more sensitive to RDX neurotoxicity than females (i.e., more convulsions; more hyperactive; greater brain weight changes). Urogenital effects have been observed in males at lower doses than in females (Levine et al., 1983b; Levine et al., 1981a, b; Cholakis et al., 1980), suggesting a possible sex-based difference in susceptibility to RDX toxicity.

Data on the incidence of convulsions and mortality from gavage studies of RDX in the rat provide some indication that pregnant animals may be a susceptible population. In the developmental toxicity study by Cholakis et al. (1980), deaths were observed in pregnant F344 rats only at a dose of 20 mg/kg-day, but convulsions were reported in a single rat at 2 mg/kg-day. In a range-finding developmental toxicity study (Angerhofer et al., 1986), mortality and convulsions were reported in pregnant Sprague-Dawley rats at a dose of ≥40 mg/kg-day, but not at ≤20 mg/kg-day, although the relatively small group sizes in this study should be noted. In the main study by these investigators, convulsions were reported in pregnant rats only at 20 mg/kg-day, but one death (in dose groups of 40 rats) was reported at both 2 and 6 mg/kg-day (Angerhofer et al., 1986). In comparison, increased mortality and convulsions were reported at ≥8 mg/kg-day in a 90-day gavage study in F344 rats (Crouse et al., 2006). The instances of one convulsion and two deaths in pregnant rats in the Cholakis et al. (1980) and Angerhofer et al. (1986) studies at doses of 2 or 6 mg/kg-day raise the possibility that pregnants animals may be more susceptible to the effects of RDX; however, direct comparison between the available gavage studies in pregnants and nonpregnant rats is uncertain because of differences in study design, including numbers of animals tested per group, test material characteristics, and rat strain. Overall, the available information is not considered sufficient to conclude that pregnant animals are a susceptible population.

There is limited evidence that CYP450 or similar enzymes are involved in the metabolism of RDX (Bhushan et al., 2003), indicating a potential for genetic polymorphisms in these metabolic enzymes to affect susceptibility to RDX. This susceptibility may also be influenced by differential expression of these enzymes during development. Individuals with epilepsy or other seizure syndromes, and in particular those that have their basis in genetic mutation to $GABA_A$ receptors, may represent another group that may be susceptible to RDX exposure. However, there is currently

no information to support predictions of how genetic polymorphisms or the presence of seizure syndromes may affect susceptibility to RDX exposure.

2. DOSE-RESPONSE ANALYSIS

2.1. ORAL REFERENCE DOSE FOR EFFECTS OTHER THAN CANCER

The oral reference dose (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. It can be derived from a no-observed-adverse-effect level (NOAEL), lowest-observed-adverse-effect level (LOAEL), or the 95% lower bound on the benchmark dose (BMDL), with uncertainty factors (UFs) generally applied to reflect limitations of the data used.

2.1.1. Identification of Studies for Dose-Response Analysis of Selected Effects

As discussed in Section 1.3.1, based on findings from oral studies in experimental animals, nervous system effects are a human hazard of hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) exposure, and kidney and other urogenital system effects are a potential human hazard of RDX exposure. There is suggestive evidence of male reproductive effects associated with RDX exposure. Although animal mortality has been reported in a number of the toxicology studies conducted for RDX, it was not considered a hazard by itself or as the basis for the derivation of a reference value. Rather, the mortality evidence was evaluated in the context of that system-specific hazard (see Sections 2.1.6 and 2.1.7 for further discussion).

The effects selected to best represent each of the hazards (see discussion in Section 1.3.1) are noted below. In order to identify the stronger studies for dose-response analysis, several attributes of the studies reporting the endpoints selected for each hazard were reviewed (i.e., study size and design, relevance of the exposure paradigm, and measurement of the endpoints of interest). In considering the study size and design, preference was given to studies using designs reasonably expected to have power to detect responses of suitable magnitude. Exposure paradigms including a route of human environmental exposure (i.e., oral and inhalation) are preferred. When developing a chronic reference value, chronic or subchronic studies are preferred over studies of acute exposure durations. Studies with a broad exposure range and multiple exposure levels are preferred to the extent that they can provide information about the shape of the exposure-response relationship. Additionally, with respect to measurement of the endpoint, studies that can reliably distinguish the presence or absence (or degree of severity) of the effect are preferred.

Human studies are generally preferred over animal studies as the basis for a reference value when quantitative measures of exposure are reported, and the reported effects are determined to be associated with exposure. The available epidemiological studies of worker populations exposed

to RDX examined the relationship between certain health endpoints and inhalation exposure; however, no epidemiological studies of ingested RDX are available. Multiple case reports support the identification of hazards associated with RDX exposure but are inadequate for dose-response analysis because they do not yield incidence estimates, exposure durations are short, and quantitative exposure information is lacking. Therefore, human studies could not be used for oral dose-response analysis or to serve as the basis for the RfD. In the absence of human data, the animal studies were considered for dose-response analysis.

Experimental animal studies considered for each health effect were evaluated using general study quality considerations discussed in Section 6 of the Preamble and in the literature search section, and the attributes described above. The rationales for selecting the strongest studies that reported the health effects are summarized below.

Nervous System Effects

Convulsions were selected for dose-response analysis as a consistent and sensitive endpoint of nervous system effects (see Section 1.3.1 for discussion). This endpoint was reported in seven studies (Crouse et al., 2006; Lish et al., 1984; Levine et al., 1983b; Levine et al., 1981a; Cholakis et al., 1980; Martin and Hart, 1974; von Oettingen et al., 1949). Table 2-1 provides an overview of the information considered in the studies reporting nervous system effects (i.e., convulsions) evaluated for dose-response analysis.

Table 2-1. Information considered for evaluation of studies that examined convulsions

Study reference	Study design and size		Relevance of exposure paradigm					Measurement of endpoint
	Design	Number of animals	Route	Duration	Number of dose groups[1]	Levels (mg/kg-d)	Purity (%)	Incidence data reported
Crouse et al. (2006)	Toxicity study	10 rats/sex/ group	Gavage	13-wk	5	4–15	99.99	Yes
Cholakis et al. (1980)	Developmental study	24–25 female rats/group	Gavage	14-d	3	0.2–20	89	Yes
Martin and Hart (1974)	Toxicity study	3 monkeys/ sex/group	Gavage	13-wk	3	0.1–10	Not specified	Yes
Levine et al. (1983b)	Toxicity and carcinogenicity bioassay	75 rats/sex/ group	Diet	2-yr	4	0.3–40	89–99	No
Lish et al. (1984)	Toxicity and carcinogenicity bioassay	85 mice/sex/ group	Diet	2-yr	4	1.5–175	89–99	No

Study reference	Study design and size		Relevance of exposure paradigm					Measurement of endpoint
	Design	Number of animals	Route	Duration	Number of dose groups[1]	Levels (mg/kg-d)	Purity (%)	Incidence data reported
Levine et al. (1981a)	Toxicity study	10 rats/sex/ group	diet	13-wk	5	10–600	85	No
von Oettingen et al. (1949)	Toxicity study	20 rats/group	diet	13-wk	3	15–50	90–97	No

[1]Excluding the control group.

Incidence of convulsions was reported in three studies of RDX—all involving gavage administration: Crouse et al. (2006), Cholakis et al. (1980) (developmental toxicity study), and Martin and Hart (1974). Qualitative findings of nervous system effects were reported in other chronic and subchronic studies—all involving dietary administration: Lish et al. (1984), Levine et al. (1983b), Levine et al. (1981a), and von Oettingen et al. (1949). Incidence data on neurotoxic effects of RDX were not collected in any of the dietary studies. For example, Levine et al. (1983b) reported only that convulsions and other nervous system effects were noted in rats exposed to RDX for 2 years at the highest dose (40 mg/kg-day) tested. The studies that included incidence data (i.e., the gavage studies) were preferred over those studies only reporting qualitative results (i.e., the dietary studies).

The three gavage studies reporting incidence data were further considered. Crouse et al. (2006) reported a dose-related increase in convulsions and tremors in both male and female F344 rats following a 90-day oral (gavage) exposure to RDX. This study used a test material of high purity and six dose groups (including the control) that provided good resolution of the dose-response curve. Cholakis et al. (1980) reported a dose-related increase in convulsions in a developmental toxicity study in F344 rats, following a 14-day exposure to RDX on gestational days (GDs) 6–19. Although this study was designed as a standard developmental toxicity study (i.e., not specifically to examine nervous system effects), it reported information on the identity of the test material and used three dose groups that adequately characterized the dose-response curve. Further, this study provided evidence of nervous system effects at a relatively low dose. The study in monkeys by Martin and Hart (1974) provides supporting evidence of nervous system effects (trembling, shaking, ataxia, hyperactive reflexes, and convulsions); however, this study was not selected for dose-response analysis because of small group sizes (n = 3/sex) and uncertainty in measures of exposures (e.g., purity of the test material was not specified, and reported emesis in some animals likely influenced the delivered dose).

Although the gavage studies reporting incidence data were preferred over four dietary studies (Lish et al., 1984; Levine et al., 1983b; Levine et al., 1981a; von Oettingen et al., 1949) that

did not provide incidence data, it is important to note that the reported neurotoxic effects in the dietary studies were observed at dose levels higher than the doses at which effects were observed in the gavage studies (Crouse et al., 2006; Cholakis et al., 1980; Martin and Hart, 1974). Given this potential difference based on dosing method, the dietary studies were also considered for quantitative analysis, despite the lack of incidence data, to evaluate the influence of oral dosing method on candidate reference values. In the 2-year study by Levine et al. (1983b), a LOAEL for nervous system effects (convulsions, tremors, and hyper-irritability) of 40 mg/kg-day and a NOAEL of 8 mg/kg-day were identified. Other studies identified higher effect levels (i.e., 100 mg/kg-day in the 2-year mouse study by Lish et al. (1984) and 50 mg/kg-day in the 3-month rat study by von Oettingen et al. (1949)), and, with the exception of Lish et al. (1984), used shorter exposure durations. The unusual dosing regimen in the Cholakis et al. (1980) 13-week mouse study precluded identification of a NOAEL and LOAEL, and the single-dose design of the 6-week dog study by von Oettingen et al. (1949) did not allow identification of a NOAEL. As discussed in Section 1.2.1 and Table 1-3, the technical report of the 13-week study by Levine et al. (1981a) inconsistently identified the dose level at which convulsions occurred; therefore, a reliable NOAEL and LOAEL from this study could not be identified.

Therefore, two gavage studies, Crouse et al. (2006) and Cholakis et al. (1980), and one dietary study, Levine et al. (1983b), were selected for dose-response analysis.

Kidney and Other Urogenital System Effects

Suppurative prostatitis was selected for dose-response analysis. It is considered to be a sensitive marker for the broader range of urogenital effects observed in F344 male rats in a 2-year study by Levine et al. (1983b). The Levine et al. (1983b) study: (1) included a histopathological examination of the kidney and other urogenital system tissues at 6-, 12-, and 24-month time points; (2) included four dose groups and a control group, and adequate numbers of animals per dose group (75/sex/group, with interim sacrifice groups of 10/sex/group at 6 and 12 months); and (3) reported individual animal data. This study, the only one to identify suppurative prostatisis, was selected for dose-response analysis.

Male Reproductive Toxicity

Testicular degeneration was selected for dose-response analysis. Lish et al. (1984) observed a dose-related increase in the incidence of testicular degeneration in mice following chronic administration of RDX in the diet. This 2-year study: (1) included histopathological examination of male reproductive organs; (2) included four dose groups and a control group, and adequate numbers of animals per dose group (85/sex/group, with interim sacrifice groups of 10/sex/group at 6 and 12 months); and (3) reported individual animal data. This study, the only one to identify testicular degeneration, was selected for dose-response analysis.

2.1.2. Methods of Analysis

No biologically based dose-response models are available for RDX. In this situation, the U.S. Environmental Protection Agency (EPA) evaluates a range of dose-response models thought to be consistent with underlying biological processes to determine how best to empirically model the dose-response relationship in the range of the observed data. Consistent with this approach, EPA evaluated dose-response information with the models available in EPA's Benchmark Dose Software (BMDS, versions 2.4 and 2.5). EPA estimated the benchmark dose (BMD) and BMDL using a benchmark response (BMR) selected for each effect. A conceptual model of the analysis approach used for RDX is provided in Figure 2-1. In this assessment, points of departure (PODs) are identified through BMD modeling (preferred) or identification of a NOAEL, and followed by animal-to-human extrapolation through the use of physiologically based pharmacokinetic (PBPK) models or the application of a dosimetric adjustment factor, depending on the data available.

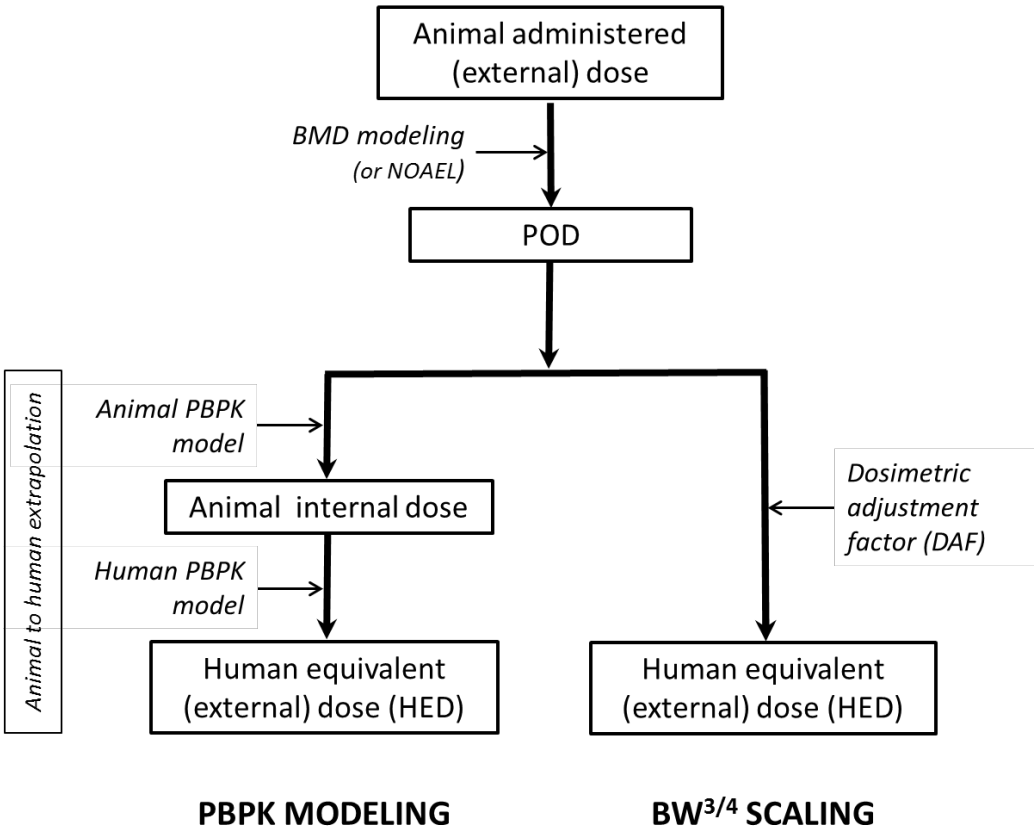


Figure 2-1. Conceptual approach to dose-response modeling for oral exposure.

Nervous System Effects

Incidence data for convulsions from Crouse et al. (2006) and Cholakis et al. (1980) were amenable to BMD modeling. For Crouse et al. (2006), statistical analysis conducted by EPA indicated no significant difference in convulsion rates of male and female rats (Mantel-Haenszel test for independence; see Table 2-2); thus, combined incidence data from male and female rats were used for modeling convulsion data from this study. A BMR of 1% extra risk (ER) for convulsions was used to address the severity of this endpoint; modeling with 5 and 10% ER is provided in Appendix D (see Section D.1.2, Tables D-3 to D-14) for comparative purposes. In general, for noncancer effects, severe endpoints are not typically used as the basis of a noncancer risk value because of relatively high uncertainty in extrapolating to a level of exposure likely to be without appreciable risk. The use of a 1% ER BMR for convulsions in Crouse et al. (2006) resulted in extrapolation below the range of the experimental doses. However, the BMD of 3.02 mg/kg-day was not far below the dose range of 4–15 mg/kg-day used in the study; thus, this extrapolation was considered moderate. In addition to uncertainty from extrapolation, model uncertainty from the use of the 1% ER BMR can be a concern. However, the BMDLs from Crouse et al. (2006) ranged from 0.54 to 2.90, a 5.4-fold difference, which is also not considered large, so the use of a 1% ER BMR did not result in substantial model uncertainty.

Because incidence data for convulsions were not provided by Levine et al. (1983b), a NOAEL was used as the POD for this dataset rather than a BMDL.

Table 2-2 summarizes the PODs derived for each data set. More detailed BMD modeling information is available in Appendix D.

Kidney/Urogenital System Effects

Incidence data on suppurative prostatitis as reported by Levine et al. (1983b) were amenable to BMD modeling. A BMR of 10% ER was applied under the assumption that it represents a minimally biologically significant level of change. Table 2-2 summarizes the POD derived using data on the incidence of suppurative prostatitis. More detailed BMD modeling information is available in Appendix D.

Male Reproductive Effects

Incidence data on testicular degeneration as reported by Lish et al. (1984) were amenable to modeling. A BMR of 10% ER was applied under the assumption that it represents a minimally biologically significant level of change. Table 2-2 summarizes the POD derived using date on the incidence of testicular degeneration. More detailed BMD modeling information is available in Appendix D.

Table 2-2. Summary of derivation of PODs following oral exposure to RDX

Endpoint and reference (exposure duration/route)	Species/sex	Model[a]	BMR	BMD (mg/kg-d)	BMDL (mg/kg-d)	POD_{HED} (mg/kg-d)		
						Administered dose[b]	RDX AUC[c]	RDX C_{max}[d]
Nervous system								
Incidence of convulsions Crouse et al. (2006) (90-d/gavage)	Male and female F344 rat, combined[e]	Multistage 2°	1% ER	3.02	0.57	0.14	0.28	0.37
Incidence of convulsions Cholakis et al. (1980) (GDs 6–19/gavage)	Female F344 rat	Quantal-linear	1% ER	0.18	0.12	0.03	0.06	0.08
Incidence of convulsions Levine et al. (1983b) (2-yr/diet)	Male and female F344 rat	LOAEL = 40 mg/kg-d; NOAEL = 8 mg/kg-d[f]				1.9	3.9	4.3
Kidney/urogenital system								
Incidence of suppurative prostatitis Levine et al. (1983b) (2-yr/diet)	Male F344 rat	LogProbit	10% ER	1.67	0.47	0.11	0.23	0.25
Male reproductive system								
Incidence of testicular degeneration Lish et al. (1984) (2-yr/diet)	Male $B6C3F_1$ mouse	LogProbit	10% ER	56.0	16.3	2.4	0.08	0.18

[a]For modeling details, see Appendix D.
[b]POD was converted to an HED using a standard DAF based on $BW^{3/4}$.
[c]POD was converted to an HED based on the equivalence of internal RDX dose (expressed as AUC for RDX concentration in arterial blood) derived using PBPK models.
[d]POD was converted to an HED based on the equivalence of internal RDX dose (expressed as peak RDX concentration in arterial blood, C_{max}) derived using PBPK models.
[e]Exact Mantel-Haenszel test for independence between convulsion incidence and sex, stratified by dose, yielded *p*-value >0.05.
[f]Nervous system effects for male and female rats reported qualitatively; incidence of convulsions and other nervous system effects was not reported. Therefore, available data do not support BMD modeling.

AUC = area under the curve; BW = body weight; DAF = dosimetric adjustment factor; ER = extra risk; HED = human equivalent dose

Human Extrapolation

EPA guidance (U.S. EPA, 2011) describes a hierarchy of approaches for deriving human equivalent doses (HEDs) from data in laboratory animals, with the preferred approach being PBPK modeling. Other approaches can include using chemical-specific information in the absence of a complete PBPK model. In lieu of either reliable, chemical-specific models or data to inform the derivation of human equivalent oral exposures, a body weight scaling to the ¾ power (i.e., $BW^{3/4}$) approach is generally applied to extrapolate toxicologically equivalent doses of orally administered agents from adult laboratory animals to adult humans.

Candidate PODs for endpoints selected from rat and mouse bioassays were expressed as HEDs. HEDs were derived using both PBPK modeling (with alternative measures of internal dose), and a $BW^{3/4}$ scaling approach. These approaches are outlined in Figure 2-1, and the resulting POD_{HED} values are presented in Table 2-2.

Extrapolation using PBPK modeling. PBPK models for RDX in rats, humans, and mice have been published (Sweeney et al., 2012a; Sweeney et al., 2012b; Krishnan et al., 2009) based on RDX-specific data. EPA evaluated and further developed these models for extrapolating doses from animals to humans (see Appendix C, Section C.1.5). In general, appropriately chosen internal dose metrics are expected to correlate more closely with toxic responses than external doses for effects that are not occurring at the point of contact (McLanahan et al., 2012). Therefore, PBPK model-derived arterial blood concentration of RDX is considered a better dose-metric for extrapolation of health effects than administered dose when there is adequate confidence in the estimated value. The PBPK models for RDX were used to estimate two dose metrics: the area under the curve (AUC) and the peak concentration (C_{max}) for RDX concentration in arterial blood. The AUC represents the average blood RDX concentration for the exposure duration normalized to 24 hours and the C_{max} represents the maximum RDX concentration for the exposure duration.

It appears logical to use RDX concentration levels in the brain as the internal dose metric for analyzing convulsion data. Nevertheless, the blood concentration of RDX was preferred as the dose metric due to greater confidence in modeling this variable. This is because of the substantially greater number of measurements of RDX blood levels used in calibrating model parameters. Additionally, predictions of RDX concentrations in the brain are highly correlated with RDX blood concentrations because the brain compartment does not have absorption, metabolism, or elimination of RDX. Greater confidence was placed in model estimates of blood AUC than peak blood concentrations because, as discussed in Appendix C, Section C.1.5, the rate constant for oral absorption (KAS) is uncertain, and peak concentrations are more sensitive to variations in this parameter than average values. RDX-induction of convulsions and seizures appears to be more strongly correlated with dose than exposure duration, which might argue for use of peak blood concentration as an appropriate dose metric; however, biological support for blood AUC, rather than peak blood concentration, comes from mechanistic information on RDX binding at the

picrotoxin convulsant site of the gamma-amino butyric acid (GABA) channel. There is evidence from examination of picrotoxin binding to $GABA_A$ that a resulting period of elevated neuronal activity post-exposure could result in increased likelihood of seizures developing over time or other longer-term effects on normal brain function (see Section 1.2.1 for further discussion). Therefore, the AUC for RDX concentration in arterial blood was selected as the internal dose metric for analyzing dose-response data for convulsions. Nevertheless, the POD_{HED} values based on both blood AUC and peak blood concentration (C_{max}) are presented in Table 2-2 for completeness.

The rodent PBPK model was applied to the BMDLs generated from BMD modeling to determine the animal internal dose, expressed as the AUC of RDX blood concentration, and representing the cross-species toxicologically equivalent (internal) dose. The human PBPK model was then applied to derive the corresponding HEDs (see Figure 2-2). Because the AUC is linear with exposure level, at least in the exposure range of interest, the value of the HED would be the same whether the rat or mouse PBPK model is applied before or after BMD modeling is performed. Because the sequence of the calculation does not influence the results, applying the PBPK model after BMD modeling is more efficient—BMD modeling would not have to be redone if there were changes to the PBPK model, and it is easier to evaluate and show two dose metrics (as discussed above). Because of relatively high confidence in the PBPK models developed for the rat and human, these models were used to derive reliable internal dose metrics for extrapolation. For datasets selected from the rat bioassays, the candidate oral values were calculated assuming cross-species toxicological equivalence of the AUC of RDX blood concentration derived from PBPK modeling. A published PBPK model for the mouse was evaluated (Sweeney et al., 2012b); however, major uncertainties were identified in this model. The mouse model was based on fitting both the absorption and metabolic rate constants to a single set of blood concentration measurements. In this study, the lowest dose at which RDX was detected was 35 mg/kg, an exposure level high enough to manifest some toxicity in the chronic mouse bioassay, and except for measurements from a single animal, all other data points were non-detects or excluded as outliers (Sweeney et al., 2012b). The type of additional data that increased confidence in the rat and human models (e.g., in vitro measurements of RDX metabolism and RDX elimination data) are not available for mice. Consequently, confidence in the mouse model parameter values is low. Further, there are no data to enable characterizing the fraction of RDX that is metabolized in the mouse; this is problematic considering evidence that indicates that the role of metabolism in RDX toxicity may differ across species (e.g., mice may have more efficient or higher expression of the cytochrome P450 [CYP450] enzymes). Given the high sensitivity of the model to the metabolic rate constant, the uncertainty in mouse toxicokinetics significantly decreases confidence in using the mouse PBPK model for predicting mouse blood RDX concentrations. (See Summary of Confidence in PBPK Models for RDX in Appendix C, Section C.1.5 for further discussion of confidence in the mouse model.) Given the low confidence in the mouse PBPK model, the preferred approach for determining candidate oral

values for endpoints selected from the mouse bioassay is that based on the administered dose of RDX extrapolated to humans using allometric $BW^{3/4}$ scaling.

Extrapolation using $BW^{3/4}$ scaling. HEDs were also calculated using a $BW^{3/4}$ scaling approach consistent with EPA guidance (U.S. EPA, 2011). PODs (BMDLs or NOAELs) based on the RDX dose administered in the experimental animal study were adjusted by a standard dosimetric adjustment factor (DAF) derived as follows:

$$DAF = (BW_a^{1/4}/BW_h^{1/4}),$$

where

BW_a = animal body weight
BW_h = human body weight

Using BW_a values of 0.25 kg for rats and 0.035 kg for mice and a BW_h of 70 kg for humans (U.S. EPA, 1988), the resulting DAFs for rats and mice are 0.24 and 0.15, respectively. Applying the DAF to the POD identified for effects in adult rats or mice yields a POD_{HED} as follows (see Table 2-2):

$$POD_{HED} = \text{laboratory animal dose (mg/kg-day)} \times DAF$$

Further details of the BMDL modeling, BMDS outputs, and graphical results for the best fit model for each dataset included in Table 2-2 can be found in Appendix D, Section D.1. Details of the PBPK model evaluation used for extrapolation from BMDL values can be found in Appendix C, Section C.1.5. Table 2-2 summarizes the results of the BMD modeling and the POD_{HED} for each data set discussed above.

2.1.3. Derivation of Candidate Values

Under EPA's *A Review of the Reference Dose and Reference Concentration Processes* (U.S. EPA, 2002) (Section 4.4.5), and as described in the Preamble, five possible areas of uncertainty and variability were considered when determining the application of UFs to the PODs presented in Table 2-2. An explanation follows:

An intraspecies uncertainty factor, UF_H, of 10 was applied to all PODs to account for potential differences in toxicokinetics and toxicodynamics in the absence of information on the variability of response in the human population following oral exposure to RDX. The available human pharmacokinetic data are not sufficient to inform human kinetic variability and derive a chemical-specific UF for intraspecies uncertainty.

An interspecies uncertainty factor, UF_A, of 3 ($10^{1/2}$ = 3.16, rounded to 3) was applied to all PODs to account for uncertainty in characterizing the toxicokinetic and toxicodynamic differences between rodents and humans. For the testicular degeneration dataset from the mouse bioassay, mouse to human extrapolation was accomplished using $BW^{3/4}$ scaling (see rationale in Section 2.1.2—Human Extrapolation), which addresses predominantly toxicokinetic and some

toxicodynamic aspects of cross-species extrapolation; residual uncertainty in toxicokinetic and toxicodynamic extrapolation remains. In the absence of chemical-specific data to quantify this uncertainty, EPA's $BW^{3/4}$ guidance (U.S. EPA, 2011) recommends use of an uncertainty factor of 3. For datasets from the rat bioassays, a PBPK model was used to convert internal doses in rats to external doses in humans (see rationale in Section 2.1.2—Human Extrapolation). This reduces toxicokinetic uncertainty in extrapolating from the rat to humans, but does not account for interspecies differences due to toxicodynamics. A UF_A of 3 was applied to account for this remaining toxicodynamic and any residual toxicokinetic uncertainty not accounted for by the PBPK model.

A subchronic to chronic uncertainty factor, UF_S, of 1 was applied to all PODs. This is because (1) in studies of subchronic or gestational exposure used to derive a POD, effects were seen at lower doses in the studies of shorter duration than in chronic studies, and (2) other studies used to derive a POD were of 2-year duration. Although EPA guidance recommends a default UF_S of 10 on the assumption that effects in a subchonic study would occur at approximately 10-fold higher concentration than in a corresponding (but absent) chronic study (U.S. EPA, 2002), the RDX database does not support a UF_S of 10. As discussed in Section 1.2.1, although Gerkin et al. (2010) introduces the possibility of effects developing over longer-term exposures to RDX, in general, seizure induction appears to be more strongly correlated with dose level than with exposure duration. The available bioassays suggest that chronic exposure would not lead to effects at lower doses than those induced by subchronic exposure. In addition, chronic dietary doses associated with convulsions were ≥35 mg/kg-day and were at least fourfold higher than gavage doses that induced convulsions in 14- and 90-day studies (i.e., 2 mg/kg-day in Cholakis et al. (1980) and 8 mg/kg-day in Crouse et al. (2006)) (also see Table 1-3 and Figure 1-1). This may be due to differences between dietary and gavage administration (see Sections 2.1.1 and 2.1.7). Nevertheless, these studies do not support the default expectation of observing effects in chronic studies at approximately 10-fold lower exposure levels than in subchronic studies. Accordingly, a UF_S of 1 was applied to PODs derived from studies of less-than-chronic duration.

A LOAEL to NOAEL uncertainty factor, UF_L, of 1 was applied to all POD values because the POD was a BMDL or a NOAEL. When the POD is a BMDL, the current approach is to address this factor as one of the considerations in selecting a BMR for BMD modeling. In this case, the BMR for modeled endpoints was selected under the assumption that the BMR represents a minimal, biologically significant change for these effects.

A database uncertainty factor, UF_D, of 3 was applied to all POD values. The oral toxicity database for RDX includes subchronic and chronic toxicity studies in the rat and mouse, a two-generation reproductive toxicity study in the rat, developmental toxicity studies in the rat and rabbit, and subchronic studies (with study design limitations) in the dog and monkey. As discussed below, some uncertainty is associated with characterization of the RDX neurotoxicity.

Analyses presented in Section 2.1.6 note that reference values derived from mortality data would be similar to the RfD for RDX based on convulsions. EPA prefers to identify reference values based on upstream (less severe) effects that would precede frank effects like convulsions and mortality. Some uncertainty remains in our understanding of RDX-induced neurotoxicity. In part, this is due to limitations in study design to assess neurotoxicity across the RDX database; the frequency of animal observations in the available studies raises concerns that there may be underreporting of the true incidence of convulsions, and in general the reporting of this effect does not include a measure of the severity at the time of observation. No follow-up studies were identified that employed more sensitive assays to assess more subtle neurotoxicity. Uncertainties in the database for RDX neurotoxicity could be addressed by:

- Analysis of "convulsions" using more detailed behavioral scoring methods. In the available studies, "convulsion" can indicate a range of observable behaviors in response to altered brain activity, ranging from involuntary limb and facial twitches to tonic-clonic seizures in which animals exhibit a sustained (seconds to hours) and widespread loss of muscle control sometimes resulting in respiratory arrest and/or death. As there are studies where convulsions occur at the same dose as mortality, the convulsive activity in these studies is interpreted as severe. Scoring methods quantifying the occurrence of different behavioral aspects of the RDX-induced convulsions, such as the Racine scale (Racine, 1972), employed in Burdette et al. (1988) would provide a much more accurate, complete, and possibly more sensitive measure of RDX neurotoxicity.
- Additional electrophysiological measures of epileptiform activity. Well-established and sensitive methods for evaluating brain activity exist. These measures could not only better describe the profile of RDX-induced convulsant activity, but could also be used to identify and quantify sub-convulsive effects of RDX exposure (e.g., EEG spiking). Electrophysiological characterization of the effects of RDX in vitro and in vivo has already been demonstrated by Williams et al. (2011). Additional studies building on this work, looking at the effects of different concentrations of RDX, could potentially identify more sensitive measures of RDX neurotoxicity.
- A FOB conducted by Crouse et al. (2006) provides information on neurobehavioral effects associated with RDX exposure, yet the results of that study did not identify notable effects associated with RDX exposure. While some components of the FOB testing conducted by Crouse et al. (2006) would be expected to give a screening-level evaluation of some stimuli-induced behaviors that have the potential to be related (e.g., response to handling, touch, click or open field), additional studies addressing whether RDX exposure alters the susceptibility to seizures elicited by traditional means could be informative. Burdette et al. (1988) examined seizure susceptibility in gavaged male Long Evans rats, but at doses ≥10 mg/kg. Further evaluation of seizure susceptibility at doses lower than 10 mg/kg, and with longer exposure durations, may identify additional measures of RDX neurotoxicity.

- Further evaluation of potential developmental neurotoxicity (and specifically seizure induction) associated with RDX exposure. Models for examining seizure-related behaviors during development exist, mainly involving manipulation and analyses in pre-weanling rodents. Hess-Ruth et al. (2007) reported possible transfer of RDX to offspring during gestation, as well as the presence of RDX in the milk of dams, indicating a potential for lactational transfer of RDX to offspring. Although examination of specific developmental neurotoxicity endpoints has not been conducted in studies of RDX toxicity, the available testing, including a two-generation reproductive toxicity study in the rat (Cholakis et al., 1980), did not report any evidence of neurobehavioral effects in offspring exposed during gestation or lactation. However, confidence in the observation is reduced as there is a question if the extent of observation in Cholakis et al. (1980) was sufficient to accurately characterize neurobehavioral effects. Additional developmental neurotoxicity studies could further rule out the possibility that RDX exposure during development might result in immediate or delayed seizure activity, or predispose animals to developing seizures as adults.

Overall, while the RDX database adequately covers major systemic effects, including reproductive and developmental effects, uncertainties in the adequacy of the database were identified in characterization of the neurotoxicity hazard. There is some concern that additional studies described above may lead to identification of a more sensitive endpoint or a lower POD. Accordingly, a UF_D of 3 was applied to all derived PODs.

Table 2-3 is a continuation of Table 2-2 and summarizes the application of UFs to each POD_{HED} to derive a candidate value for each data set. The candidate values presented in Table 2-3 are preliminary to the derivation of the organ/system-specific reference values. These candidate values are considered individually in the selection of a representative oral reference value for a specific hazard and subsequent overall RfD for RDX.

Table 2-3. Effects and corresponding derivation of candidate values

Endpoint and reference	POD_{HED}[a]	POD type	UF_A	UF_H	UF_L	UF_S	UF_D	Composite UF	Candidate value (mg/kg-d)
Nervous system (rat)									
Incidence of convulsions Crouse et al. (2006)	0.28	$BMDL_{01}$	3	10	1	1	3	100	2.8×10^{-3}
Incidence of convulsions Cholakis et al. (1980)	0.06	$BMDL_{01}$	3	10	1	1	3	100	6.0×10^{-4}
Incidence of convulsions Levine et al. (1983b)	3.9	NOAEL	3	10	1	1	3	100	3.9×10^{-2}
Kidney/urogenital system (rat)									
Incidence of prostate suppurative inflammation Levine et al. (1983b)	0.23	$BMDL_{10}$	3	10	1	1	3	100	2.3×10^{-3}
Male reproductive system (mouse)									
Incidence of testicular degeneration Lish et al. (1984)	2.4	$BMDL_{10}$	3	10	1	1	3	100	2.4×10^{-2}

[a]POD_{HED} values based on data from the rat were derived using PBPK modeling; the POD_{HED} based on data from the mouse was derived using $BW^{3/4}$ adjustment (see Section 2.1.2 and discussion of the PBPK models above and in Appendix C, Section C.1.5).

Figure 2-2 presents graphically the candidate values, UFs, and POD_{HED} values, with each bar corresponding to one data set described in Tables 2-2 and 2-3.

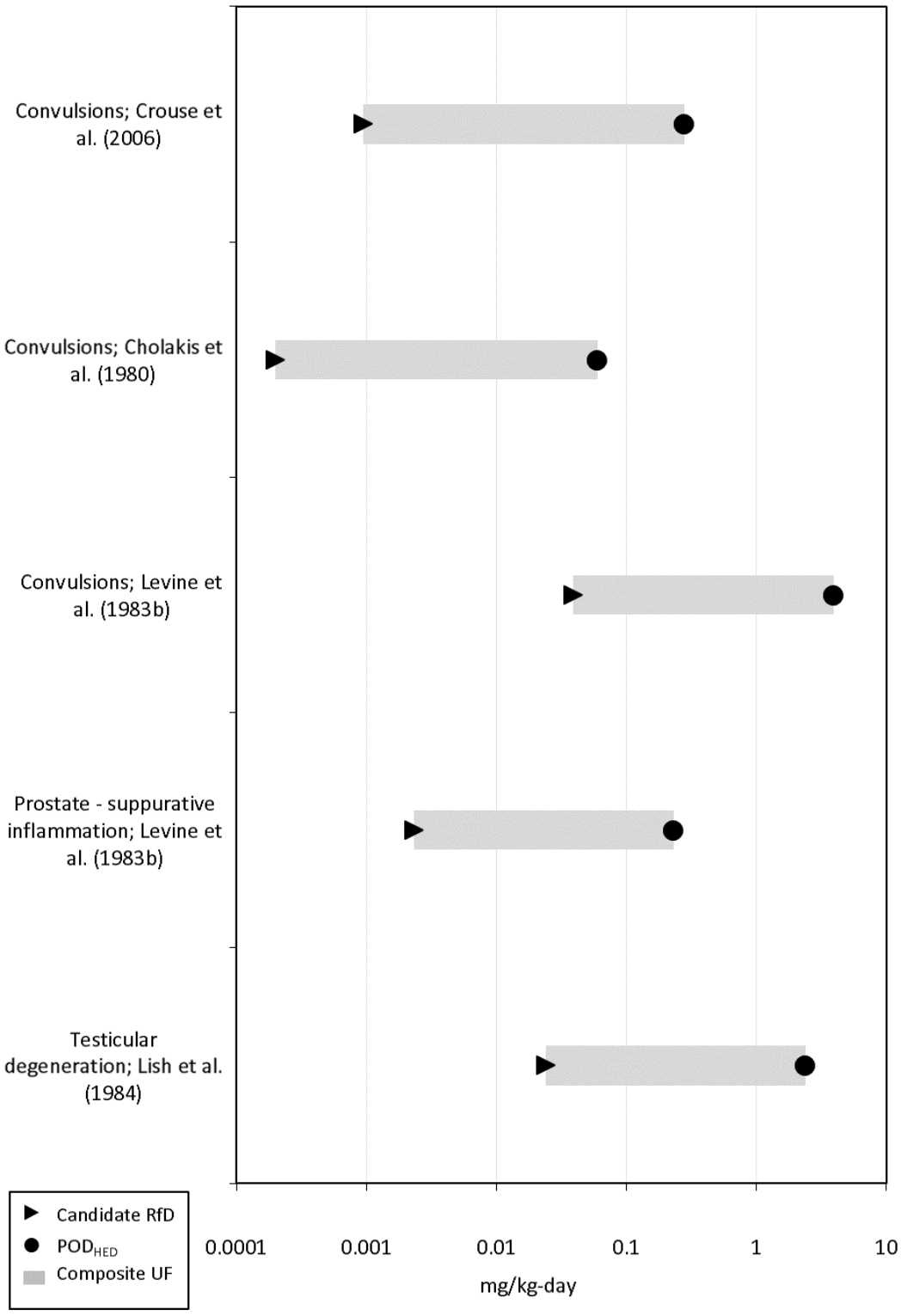


Figure 2-2. Candidate values with corresponding POD and composite UF.

2.1.4. Derivation of Organ/System-Specific Reference Doses

Table 2-4 distills the candidate values from Table 2-3 into a single value for each organ or system. Organ- or system-specific reference values may be useful for subsequent cumulative risk assessments that consider the combined effect of multiple agents acting at a common site.

Table 2-4. Organ/system-specific RfDs and overall RfD for RDX

Effect	Basis	RfD (mg/kg-d)	Study exposure description	Confidence
Nervous system	Incidence of convulsions (Crouse et al., 2006)	3×10^{-3}	Subchronic	Medium
Kidney/urogenital system	Incidence of suppurative prostatitis (Levine et al., 1983b)	2×10^{-3}	Chronic	Low
Male reproductive system	Incidence of testicular degeneration (Lish et al., 1984)	2×10^{-2}	Chronic	Low
Overall RfD	**Nervous system**	**3×10^{-3}**	**Subchronic**	**Medium**

Nervous System Effects

The organ/system-specific RfD for nervous system effects was based on the incidence of convulsions in F344 rats reported in Crouse et al. (2006), a well-conducted study that used a 99.99% pure form of RDX, five closely-spaced dose groups that provided a good characterization of the dose-response curve for convulsions, and an endpoint (convulsions) that was replicated across multiple studies. Although the candidate value derived from the developmental toxicity study in F344 rats by Cholakis et al. (1980) is lower (by approximately fivefold), there is greater certainty in the value derived from Crouse et al. (2006). Crouse et al. (2006) was specifically designed to assess the nervous system effects of RDX (including a functional observational battery), whereas Cholakis et al. (1980) was designed as a developmental toxicity study with only routine monitoring of clinical signs (the methods section states that "Dams were monitored daily for toxic signs"). Crouse et al. (2006) used five dose groups (plus the control) that provided good resolution of the dose-response curve for RDX-induced convulsions, whereas Cholakis et al. (1980) used only three dose group (plus the control) with order of magnitude dose spacing, resulting in a less well defined characterization of the dose-response curve for this endpoint. Further, Crouse et al. (2006) used a higher purity test material than did Cholakis et al. (1980) (99.99% versus 88.6%, respectively). Finally, the Crouse et al. (2006) study used a longer exposure duration (90 days) than did the Cholakis et al. (1980) study (14 days), and is more representative of a chronic exposure duration. The lower candidate reference value from the Cholakis et al. (1980) developmental toxicity study could indicate that pregnant animals are a susceptible population, which could support selection of

this study as the basis for the RfD; however, as discussed in Section 1.3.3, the available studies in pregnant and nonpregnants rats cannot be directly compared, and the available information is not considered sufficient to identify pregnant animals as a susceptible population.

As discussed in Section 2.1.1, the 2-year dietary study by Levine et al. (1983b) was also considered for RfD derivation because the available oral studies suggest that bolus doses of RDX received with gavage administration may induce nervous system effects at doses lower than those resulting from dietary administration (recognizing that differences in particle size and purity of the test material may confound direct comparisons between gavage and dietary administration). Convulsion data from Levine et al. (1983b) yielded a POD_{HED} 14-fold higher than the POD_{HED} derived from Crouse et al. (2006). The POD derived from the Levine et al. (1983b) study is considered less certain than that derived from Crouse et al. (2006). Levine et al. (1983b) did not provide information on the incidence of neurotoxic effects, and BMD analysis was thus not supported (i.e., the POD was based on a NOAEL). As discussed in Section 1.2.1, the frequency of daily observations in the Levine et al. (1983b) study may not have been sufficient to provide an accurate measure of the occurrence of nervous system effects, potentially leading to underestimation of convulsions and other nervous system effects. For these reasons, and in light of the fact that data from the Levine et al. (1983b) study yielded a higher POD, Levine et al. (1983b) was not used as the basis for the organ/system-specific RfD for nervous system effects.

Kidney/Urogenital Effects

A single data set for incidence of suppurative prostatitis in male F344 rats as reported in a 2-year dietary study by Levine et al. (1983b) was brought forward for quantitative analysis as a sensitive marker for the broader array of RDX-associated effects observed in the urogenital system. The RfD for kidney and other urogenital effects is based on this dataset.

Male Reproductive Effects

A single dataset for male reproductive effects, specifically the incidence of testicular degeneration as reported in male $B6C3F_1$ mice exposed to RDX in diet for 24 months (Lish et al., 1984), was brought forward for quantitative analysis. The RfD for male reproductive effects is based on this dataset.

2.1.5. Selection of the Overall Reference Dose

Multiple organ/system-specific reference doses were derived for effects identified as potential hazards from RDX exposure, including nervous system effects, kidney and other urogenital effects, and male reproductive effects. Evidence for nervous system effects, and specifically convulsions, was observed in multiple studies, in multiple species, and following a range of exposure durations. In addition, the organ/system-specific RfD for nervous system effects was the lowest among the organ/system-specific RfDs derived for RDX. Evidence for dose-related effects on the urogenital system comes primarily from a single 2-year toxicity study in male rats

(Levine et al., 1983b), and evidence for male reproductive effects comes primarily from a single 2-year toxicity study in mice (Lish et al., 1984); neither a second chronic study in the rat that evaluated prostate histopathology nor a second mouse study was available to validate and replicate these findings.

The organ/system-specific RfD of **3×10^{-3} mg/kg-day** for nervous system effects in the rat as reported by Crouse et al. (2006) is selected as the overall RfD for RDX given the strength of evidence for the nervous system as a hazard of RDX exposure, and as the lowest organ/system-specific RfD. This overall RfD provides an exposure level below which effects associated with RDX exposure are not expected to occur.

The overall RfD is derived to be protective of all types of effects for a given duration of exposure, and is intended to protect the population as a whole, including potentially susceptible subgroups (U.S. EPA, 2002). Decisions concerning averaging exposures over time for comparison with the RfD should consider the types of toxicological effects and specific lifestages of concern. Fluctuations in exposure levels that result in elevated exposures during these lifestages could potentially lead to an appreciable risk, even if average levels over the full exposure duration were less than or equal to the RfD. In the case of RDX, no specific lifestages have been identified as a potentially susceptible subgroup.

2.1.6. Comparison with Mortality LD_{01}s

As previously discussed, mortality was considered in discussions of other organ/system-specific toxicity (and in particular, effects on the nervous system and kidney). EPA did not develop a candidate RfC from mortality because EPA generally does not develop reference values based on frank effects such as mortality, rather, reference values are generally based on earlier (less severe) upstream events, where possible, in order to protect against all adverse outcomes. Nevertheless, additional analysis of mortality data was undertaken because some studies (see Table 2-5) identified mortality at the same RDX dose that induced nervous system effects (Crouse et al. (2006); Angerhofer et al. (1986); Cholakis et al. (1980); von Oettingen et al. (1949)).

Table 2-5. Comparison of dose levels associated with mortality and convulsions in selected studies

Study	Doses associated with mortality	Doses associated with convulsions
Crouse et al. (2006) Rats, F344, 10/sex/group 0, 4, 8, 10, 12, or 15 mg/kg-d 13 wks/gavage	≥8 mg/kg-d	≥8 mg/kg-d
von Oettingen et al. (1949) Rats, sex/strain not specified, 20/group 0, 15, 25, or 50 mg/kg-d 13 wks/diet	≥25 mg/kg-day	≥25 mg/kg-d
Cholakis et al. (1980) Rats, F344, 24–25 females/group 0, 0.2, 2.0, or 20 mg/kg-d GDs 6–19/gavage	20 mg/kg-d	Primarily 20 mg/kg-d; 1 convulsion at 2 mg/kg-d
Angerhofer et al. (1986) Rats, Sprague-Dawley, 39–51 mated females/group 0, 2, 6, or 20 mg/kg-d GDs 6–15/gavage	Primarily at 20 mg/kg-d, but one death each at 2 and 6 mg/kg-d	20 mg/kg-d

A discussion of mortality evidence for RDX is presented in Appendix C, Section C.3.1, and the relationship between mortality and nervous system effects in Sections 1.2.1 and 1.3.1. Unscheduled deaths were observed as early as day 8 of a 90-day gavage study (Crouse et al., 2006) and in development toxicity studies with exposure durations of two weeks (Angerhofer et al. (1986); Cholakis et al. (1980)).

Given the proximity in the dose at which mortality and nervous system effects were observed in several studies, the dose-response relationships for mortality were compared across studies with similar durations to those in Table 2-5 by comparing the LD_{01} (the dose expected to be lethal to 1% of the animals) or NOAELs derived from each study. In addition, these LD_{01}s and NOAELs were compare to the BMD_{01} for convulsions used to derive the RfD.[14] Interpretation of mortality data from chronic exposure studies in mice and rats is complicated by other treatment-related effects and pathology regularly observed in aging animals (e.g., kidney pathology, neoplastic lesions), and was not considered in this analysis. Other studies that were less informative and not considered in this analysis are not presented in Table 2-6.[15]

[14] BMDs were compared, as opposed to BMDLs, because, as stated on p. 20 of the BMD Technical Guidance (U.S. EPA, 2012b), "In general, it is recommended that comparisons across chemicals/studies/endpoints be based on central estimates; this is in contrast to using lower bounds for PODs for reference values..."

[15]The following less informative studies were not included in the analysis of early mortality:

Table 2-6. Summary of dose-response evaluation for mortality following oral exposure to RDX

Reference (exposure duration/route)	Species/sex	Model[a]	BMR	LD_{01} (mg/kg-d)	LDL_{01} (mg/kg-d)
Diet studies					
Lish et al. (1984) (11-week data from 2-yr study/diet)	Male and female $B6C3F_1$ mouse	Not amenable to modeling	NOAEL: 35 mg/kg-day 95% CI for response: 0-4%		
Levine et al. (1981a) (13-wk/diet)	Male and female F344 rat, combined	Multistage 4°	1% ER	7.8	2.2
von Oettingen et al. (1949) (13-wk/diet)	Rats, sex/strain not specified	Not amenable to modeling	NOAEL: 15 mg/kg-day 95% CI for response: 0-15%		
Cholakis et al. (1980) (2-generation design/diet)	Female CD rat	Not amenable to modeling	NOAEL: 16 mg/kg-day 95% CI for response: 0–13%		
Levine et al. (1983b) (13-week data from 2-yr study/diet)	Male and female F344 rat	NA (no mortality at highest dose tested)	NOAEL: 40 mg/kg-day 95% CI for response: 0-4%		
Cholakis et al. (1980) (13-wk/diet)	Male and female F344 rat	NA (no mortality at highest dose tested)	NOAEL: 40 mg/kg-day 95% CI for response: 0-25%		
Gavage studies					
Crouse et al. (2006) (90-d/gavage)	Male and female F344 rat, combined	Multistage 2°	1% ER	2.1	0.46
Cholakis et al. (1980) (GDs 6–19/gavage)	Female F344 rat	Not amenable to modeling	NOAEL: 2 mg/kg-day 95% CI for response: 0–12%		
Angerhofer et al. (1986) (GD 6–15/gavage)	Female SD rat	Multistage 3°	1% ER	1.7	0.59
Cholakis et al. (1980) (GDs 7–29/gavage)	Female New Zealand white rabbit	NA (no mortality at highest dose tested)	NOAEL: 20 mg/kg-day 95% CI for response: 0-22%		

- 13-week dietary study in the mouse by Cholakis et al. (1980). Mortality was observed only in the high-dose group (257–276 mg/kg-day TWA), and the unusual dosing regimen precluded identification of a NOAEL or LOAEL.
- 13-week dietary study in the dog by Hart (1974) and 13-week study in the monkey by Martin and Hart (1974). Both studies used small group sizes (3 animals/dose group), and no animals died on study (although one high-dose monkey was euthanized).
- 6-week dietary study in the dog from the 1949 publication by von Oettingen et al. (1949). This dog study included only one treatment group and recorded only one death.
- 30-day gavage study in the rat by MacPhail et al. (1985). The authors did not identify treatment-related mortality, but reporting was limited.

Reference (exposure duration/route)	Species/sex	Model[a]	BMR	LD_{01} (mg/kg-d)	LDL_{01} (mg/kg-d)

[a]For modeling details, see Appendix D, Section D.1.3.
ER = extra risk
LD_{01} = dose expected to be lethal to 1% of the animals
LDL_{01} = lower confidence limit on the LD_{01}

Of the studies in Table 2-6, dose-response analysis was conducted for all studies that showed an increased incidence of unscheduled deaths. LD values are provided in Table 2-6, and detailed modeling results are provided in Appendix D, Section D.1.3. Mortality was observed only at the highest dose tested at week 11 in the 2-year mouse study by Lish et al. (1984), in the 13-week rat study by von Oettingen et al. (1949), and in the two-generation reproductive and developmental toxicity studies by Cholakis et al. (1980). In these cases, data were not amenable to LD_{01} estimation, and a NOAEL (with its associated confidence interval, CI) was used in this comparative analysis instead.

LD_{01} values for mortality in Table 2-6 range from 1.7 mg/kg-day (10-day gavage exposure in pregnant rats) to 7.8 mg/kg-day (13-week dietary exposure in rats), with the lower values generally from studies that administered RDX by gavage. These values may be compared to the BMD for convulsions from Crouse et al. (2006) that was used as basis for the overall RfD for RDX. The BMD for convulsions of 3.0 mg/kg-day is in the middle of the distribution of calculated LD_{01}s, and the lowest LD_{01} of 1.7 mg/kg-day is within twofold of the convulsion BMD of 3.0 mg/kg-day.

The NOAELs from studies where mortality was observed tend to be higher than the LDs. However, NOAELs are not directly comparable to BMD_{01}s for several reasons. Confidence intervals for the observed responses of 0% characterize some statistical uncertainty for NOAELs from studies that could not be modeled (note that the upper bound on an observed response of 0% is not directly comparable to a lower bound on a benchmark dose). The confidence intervals suggest that comparable 1% levels for these datasets could be somewhat lower than the NOAELs. On the other hand, dose-spacing can affect the interpretation of NOAELs, such that from the Cholakis et al. (1980) developmental toxicity study because of the wide (order-of-magnitude) spacing between doses in that study (i.e., the reported NOAEL is 10-fold lower than the dose associated with 17% mortality at 20 mg/kg-day).

In general, this comparison indicates that reference values derived from mortality data would be similar to the final RfD for RDX based on convulsions, assuming the application of the same extrapolation procedures and uncertainty factors. However, the similarity in RDX doses associated with both mortality and nervous system effects should be taken into consideration when using this RfD, and in particular in evaluating exposures that exceed the RfD in light of the severity of mortality.

2.1.7. Uncertainties in the Derivation of the Reference Dose

To derive the RfD, the UF approach (U.S. EPA, 2000a, 1994) was applied to a POD_{HED} based on nervous system effects in rats exposed to RDX for a subchronic duration. UFs were applied to the POD_{HED} values to account for uncertainties in extrapolating from an animal bioassay to human exposure, the likely existence of a diverse human population of varying susceptibilities, and subchronic to chronic duration. For the most part, these extrapolations are carried out with default approaches given the lack of data to inform individual steps. One exception is the use of PBPK modeling to perform interspecies (i.e., rat to human) extrapolation. Uncertainties associated with the PBPK models are considered in Appendix C, Section C.1.5.

Nervous system effects have been documented in multiple studies and animal species and strains; however, some uncertainty is associated with the incidence of reported neurological effects in studies that employed a study design that did not monitor animals with sufficient frequency to accurately record neurobehavioral effects, including convulsions. In the study used to derive the RfD (Crouse et al., 2006), Johnson (2015a) noted that convulsions were observed infrequently outside the dosing period; more often, seizures were observed during the 2-hour (gavage) dosing period, typically within 60–90 minutes of dosing. Similar information was not available for other studies to assess the likelihood that observations of convulsions were missed. However, animals were not monitored continuously during the Crouse et al. (2006) study, and investigators reported that nearly all observed pre-term deaths in rats exposed to the three higher doses were preceded by signs of neurotoxicity. If an animal died during the study as a result of effects on the nervous system, convulsions preceding death could have been missed, resulting in an underestimation of the incidence of convulsions. Conversely, attributing all mortality to neurotoxicity (i.e., all deaths were preceded by convulsions that may not have been observed) could result in an overestimation of the incidence of convulsions. A dose-response analysis of the combined incidence of seizures and mortality from Crouse et al. (2006) was conducted to evaluate the impact of these assumptions, as the true convulsion incidence would likely fall somewhere between the observed convulsion incidence and the combined incidence of convulsions and mortality. The POD_{HED} of 0.24 mg/kg-day for a combined incidence of convulsions and mortality[16] was compared to the POD_{HED} of 0.28 mg/kg-day for convulsions alone, indicating that the addition of mortality incidence did not have a significant impact. Therefore, the RfD based on the incidence of convulsions alone does not appear to underestimate the toxicity associated with RDX.

Some uncertainty is also associated with the influence of the method of oral dosing on the magnitude of dose required to induce nervous system effects. As noted in Section 1.2.1, gavage

[16]BMD = 2.56 mg/kg-day; BMDL = 0.49 mg/kg-day (see Appendix D.1.2 for BMD modeling results). The POD_{HED} value was derived using PBPK modeling (see Section 2.1.2 and discussion of the PBPK models in Appendix C, Section C.1.5).

administration generally induced convulsions in experimental animals at lower doses than did dietary administration, possibly due to the bolus dose resulting from gavage administration that could lead to comparatively faster absorption and higher peak blood concentrations of RDX. To some extent, this uncertainty is reflected in the 14-fold difference in the candidate POD_{HED} values derived from the Crouse et al. (2006) (gavage administration) and Levine et al. (1983b) (dietary administration) studies. A more rigorous examination of the effect of oral dosing method cannot be performed because of the differences in test materials and study designs used in the available gavage and dietary studies that could also have contributed to differences in response (e.g., test article purity and particle size, number and spacing of dose groups, exposure duration, frequency of clinical observations, and thoroughness of the reporting of observations).

Although the database is adequate for reference value derivation, uncertainty is associated with the consistency in toxicity results across studies that used RDX test materials that differed in purity, formulation, and particle size. There is evidence that differences in test material formulation and particle size (i.e., the increased surface area associated with finely powdered RDX allows for increased absorption) can affect oral bioavailability of RDX and subsequent toxicity (see discussion in Appendix C, Section C.1.5, Absorption of RDX from the GI Tract).

2.1.8. Confidence Statement

A confidence level of high, medium, or low is assigned to the study used to derive the RfD, the overall database, and the RfD itself, as described in Section 4.3.9.2 of EPA's *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry* (U.S. EPA, 1994). The overall confidence in this RfD is medium. Confidence in the principal study (Crouse et al., 2006) is high. The study was well-conducted, utilized 99.99% pure RDX, and had five closely-spaced dose groups that allowed characterization of dose-response curves for convulsions in the dose range of interest. One limitation identified by study authors was the limited ability of the FOB to fully identify neurobehavioral effects at doses ≥8 mg/kg-day due to the timing of the dosing procedure and timing of the FOB screening. Confidence in the database is medium. The database includes three chronic studies in rats and mice; eight subchronic studies in rats, mice, dogs, and monkeys; two short-term studies; and four reproductive/developmental toxicity studies in rats and rabbits (including a two-generation reproductive study). Confidence in the database is reduced largely because of (1) differences in test material used across studies, and (2) uncertainties in the influence of oral dosing methods. As discussed in Section 2.1.7 and Appendix C, Section C.1.5, differences in test material formulation and particle size may affect RDX absorption and subsequent toxicity, which in turn could influence the characterization and integration of toxicity findings across studies. The available evidence also suggests that bolus dosing of RDX that results from gavage administration induces neurotoxicity at doses lower than administration in the diet, although a rigorous examination of these differences cannot be performed with the available database. To the extent that dietary administration is more representative of potential human exposures to RDX, the use of toxicity data from a gavage (bolus dosing) study introduces

uncertainty in the RfD. Reflecting high confidence in the principal study and medium confidence in the database, overall confidence in the RfD is medium.

2.1.9. Previous IRIS Assessment

The previous RfD for RDX, posted to the Integrated Risk Information System (IRIS) database in 1988, was based on a 2-year rat feeding study by Levine et al. (1983b). The no-observed-effect level (NOEL) of 0.3 mg/kg-day based on suppurative inflammation of the prostate in male F344 rats from this study was identified as the POD. An RfD of 3×10^{-3} mg/kg-day was derived following application of an overall UF of 100 ($UF_A = 10$, $UF_H = 10$).

2.2. INHALATION REFERENCE CONCENTRATION FOR EFFECTS OTHER THAN CANCER

The inhalation reference concentration (RfC, expressed in units of mg/m^3) is defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It can be derived from a NOAEL, LOAEL, or the 95% lower bound on the benchmark concentration (BMCL), with UFs generally applied to reflect limitations of the data used.

As discussed in Section 1.3.1, the available inhalation literature does not support characterization of the health hazards specifically associated with chronic inhalation exposure to RDX, nor do the studies support quantitative dose-response analysis. Of the available human epidemiological studies of RDX (West and Stafford, 1997; Ma and Li, 1993; Hathaway and Buck, 1977), none provided data that could be used for dose-response analysis. The studies by Ma and Li (1993) of neurobehavioral effects in Chinese workers and West and Stafford (1997) of hematological abnormalities in ordnance factory workers had numerous methodological limitations that preclude their use for quantitative analysis (see Literature Search Strategy | Study Selection and Evaluation). The study by Hathaway and Buck (1977) found no evidence of adverse health effects in munition plant workers (based on evaluation of liver function, renal function, and hematology), and therefore does not identify a POD at which there would be an effect from which to derive an RfC. Multiple case reports provide some evidence of effects in humans associated with acute exposure to RDX; however, while case reports can support the identification of hazards associated with RDX exposure, data from case reports are inadequate for dose-response analysis and subsequent derivation of a chronic reference value because of short exposure durations and incomplete or missing quantitative exposure information.

As discussed in Literature Search Strategy | Study Selection and Evaluation, a single experimental animal study involving inhalation exposure was identified in the Defense Technical Information Center (DTIC) database; the study is not publicly available. However, the study would

not have provided useful data on responses to inhaled RDX, as it was limited by small numbers of animals tested, lack of controls, and incomplete reporting of exposure levels.

Therefore, the available health effects literature does not support the derivation of an RfC for RDX. While inhalation absorption of RDX particulates is a plausible route of exposure, there are no toxicokinetic studies of RDX inhalation absorption to support an inhalation model. Therefore, a PBPK model for inhaled RDX was not developed to support route-to-route extrapolation from the RfD.

2.2.1. Previous IRIS Assessment

An RfC for RDX was not previously derived under the IRIS Program.

2.3. ORAL SLOPE FACTOR FOR CANCER

The oral slope factor (OSF) is a plausible upper bound on the estimate of risk per mg/kg-day of oral exposure. The OSF can be multiplied by an estimate of lifetime exposure (in mg/kg-day) to estimate the lifetime cancer risk.

2.3.1. Analysis of Carcinogenicity Data

As noted in Section 1.3.2, there is "suggestive evidence of carcinogenic potential" for RDX. The *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a) state:

> When there is suggestive evidence, the Agency generally would not attempt a dose-response assessment, as the nature of the data generally would not support one; however, when the evidence includes a well-conducted study, quantitative analyses may be useful for some purposes, for example, providing a sense of the magnitude and uncertainty of potential risks, ranking potential hazards, or setting research priorities.

In the case of RDX, there are well-conducted studies that tested large numbers of animals at multiple dose levels (Lish et al., 1984; Levine et al., 1983b), making the cancer response suitable for dose-response analysis. Considering the data from these studies, along with the uncertainty associated with the suggestive nature of the weight of evidence, quantitative analysis of the tumor data may be useful for providing a sense of the magnitude of potential carcinogenic risk.

The incidences of liver and lung tumors in female mice from the study by Lish et al. (1984) were selected for quantitative dose-response analysis. The study by Lish et al. (1984): (1) included comprehensive histopathological examination of major organs; (2) contained four dose groups and a control; (3) used adequate numbers of animals per dose group (85/sex/group, with interim sacrifice groups of 10/sex/group at 6 and 12 months) and a sufficient overall exposure duration (2 years); and (4) adequately reported methods and results (including individual animal data). Female mouse liver tissues from the original unpublished study by Lish et al. (1984) were reevaluated by a pathology working group (PWG) (Parker et al., 2006) in order to apply more up-to-date histopathological criteria established by Harada et al. (1999). The updated liver tumor

incidences from the PWG reanalysis of Lish et al. (1984) were used for quantitative dose-response analysis.

In the case of both liver and lung tumors, benign and malignant tumors (i.e., adenomas and carcinomas) were combined for dose-response analysis because benign and malignant tumors in both organs develop from the same cell line and there is evidence for progression from benign to the malignant stage (U.S. EPA, 2005a; McConnell et al., 1986). Female mouse liver and lung tumor incidences from the Lish et al. (1984) study are summarized in Appendix D, Table D-23.

The incidence of hepatocellular carcinomas in male F344 rats from the study by Levine et al. (1983b) and the incidence of alveolar/bronchiolar carcinomas in male B6C3F$_1$ mice from the study by Lish et al. (1984) were also considered for quantitative dose-response analysis. Both studies were well conducted, using similar study designs (described above). In both instances, the response was less robust than the response observed in female mice from the Lish et al. (1984) study. The hepatocellular carcinoma result in male F344 rats is based on a small number of tumors (two each at 8 and 40 mg/kg-day, and one in controls). The alveolar/bronchiolar carcinomas in male B6C3F$_1$ mice showed a positive trend; however, a positive trend was not observed when the incidence of adenomas and carcinomas was combined. Modeling results are provided in Appendix D, Section D.2.3 for comparison.

2.3.2. Dose-Response Analysis—Adjustments and Extrapolation Methods

The EPA *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a) recommend that the method used to characterize and quantify cancer risk from a chemical be determined by what is known about the mode of action (MOA) of the carcinogen and the shape of the cancer dose-response curve. The linear approach is recommended when there are MOA data to indicate that the dose-response curve is expected to have a linear component below the POD or when the weight-of-evidence evaluation of all available data are insufficient to establish the MOA for a tumor site (U.S. EPA, 2005a). In the case of RDX, the mode of carcinogenic action for hepatocellular and alveolar/bronchiolar tumors is unknown (see discussion of Mechanistic Evidence in Section 1.2.5). Therefore, a linear low-dose extrapolation approach was used to estimate human carcinogenic risk associated with RDX exposure.

The survival curves were compared across dose groups in each study to determine whether time of death should be incorporated in the dose-response analysis of tumors. For female mice in Lish et al. (1984), the survival curves were determined to be similar across dose groups after the dose was reduced in the high-dose group to 100 mg/kg-day (log-rank test, p-value ≥0.10); therefore, a time-to-tumor analysis was not necessary for this study. Tumor incidence was modeled using the multistage-cancer models in BMDS (versions 2.4 and 2.5). A standard BMR of 10% ER was applied to both tumor sites in the mouse.

Given the finding of an association between RDX exposure in the female mouse and increased tumor incidence at two tumor sites, basing the OSF on only one tumor site could potentially underestimate the carcinogenic potential of RDX. Therefore, an analysis that combines

the results from the mouse liver and lung tumor incidence is preferred. The MS-COMBO procedure (BMDS, version 2.5) extends the multistage-cancer models to the case with multiple tumors assuming independence between tumor types. There is no known biological relationship between liver and lung tumors in RDX-exposed mice, and therefore, as noted by the National Research Council (NRC, 1994), this assumption of independence is considered not likely to produce substantial error in risk estimates. MS-COMBO analyzes tumor incidence as present if either organ (or both) has a tumor and as absent otherwise. The procedure derives a maximum likelihood estimate of the combined risk at a 95% confidence level based on the parameter values obtained for the individual tumor multistage model fits.

EPA's preferred approach for extrapolating results from animal studies to humans is toxicokinetic modeling. As described in Appendix C, Section C.1.5, PBPK models for RDX in mice and humans published by Sweeney et al. (2012b) were evaluated and further developed by EPA. Consideration was given to whether the available toxicokinetic information supported using an internal dose metric derived by PBPK modeling. The available mechanistic data (Section 1.2.5) point to some evidence, although not conclusive, that RDX-generated metabolites may be implicated in the observed tumorigenicity in the female mouse. However, there are no data on the toxicokinetics of RDX metabolites, and metabolism in the liver is the only route of elimination of RDX in the PBPK model. In this case, as is to be expected from mass balance principles, the PBPK modeling provides no further information; the HED obtained from the model-estimated amount of total RDX metabolites scaled by $BW^{3/4}$ was equal to that calculated using administered dose scaled by $BW^{3/4}$. In addition to the lack of data on metabolism, other major uncertainties were identified in the mouse PBPK modeling; EPA's evaluation of these uncertainties is discussed in Summary of Confidence in PBPK Models for RDX in Appendix C, Section C.1.5. Therefore, the PBPK model developed for the mouse was not used, and consistent with the EPA's *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), the preferred approach for calculating an HED from the mouse tumors is adjustment of the administered dose by allometric scaling to achieve toxicological equivalence across species.

As discussed in Section 2.1.1, the administered dose in animals was converted to an HED on the basis of $(\text{body weight})^{3/4}$ (U.S. EPA, 1992). This was accomplished by multiplying administered dose by $(\text{animal body weight in kg/human body weight in kg})^{1/4}$ (U.S. EPA, 1992), where the body weight for the mouse is 0.035 kg and the reference body weight for humans is 70 kg (U.S. EPA, 1988). Details of the BMD modeling can be found in Appendix D, Section D.2.

2.3.3. Derivation of the Oral Slope Factor

The lifetime cancer OSF for humans is defined as the slope of the line from the BMR (10% ER) at the BMDL to the estimated control response at zero ($OSF = 0.1/BMDL_{10\text{-}HED}$). This slope, a 95% upper confidence limit on the true slope, represents a plausible upper bound on the true slope or risk per unit dose. The PODs estimated for each mouse tumor site are summarized in Table 2-7.

Using linear extrapolation from the $BMDL_{10\text{-HED}}$, human equivalent OSFs were derived for each tumor site individually and both sites combined and are listed in Table 2-7.

Table 2-7. Model predictions and OSFs for hepatocellular and alveolar/ bronchiolar adenomas or carcinomas in female B6C3F$_1$ mice administered RDX in the diet for 2 years (Lish et al., 1984)

Tumor type	Selected model	BMR	BMD (mg/kg-d)	BMDL (mg/kg-d)	$BMD_{10\text{-HED}}$[a] (mg/kg-d)	POD = $BMDL_{10\text{-HED}}$[b] (mg/kg-d)	OSF[c] $(mg/kg\text{-}d)^{-1}$
Hepatocellular adenomas or carcinomas[d]	Multistage 1°	10% ER	64.2[e]	32.6[e]	9.56	4.89	0.020
Alveolar/bronchiolar adenomas or carcinomas	Multistage 1°	10% ER	52.8	27.7	7.92	4.16	0.024
Liver + lung tumors	Multistage 1° (MS-COMBO)	10% ER	29.0[e]	17.7[e]	4.35	2.66	0.038

[a]$BMD_{10\text{-HED}} = BMD_{10} \times (BW_a^{1/4}/BW_h^{1/4})$, where BW_a = 0.035 kg, and BW_h = 70 kg.
[b]$BMDL_{10\text{-HED}} = BMDL_{10} \times (BW_a^{1/4}/BW_h^{1/4})$, where BW_a = 0.035 kg, and BW_h = 70 kg.
[c]OSF = BMR/$BMDL_{10\text{-HED}}$, where BMR = 0.1 (10% ER).
[d]Incidences of female mouse liver tumors from Lish et al. (1984) are those reported in the PWG reevaluation (Parker et al., 2006).
[e]Data for hepatocellular adenomas and carcinomas and for liver and lung tumors combined were remodeled using the original sample sizes provided in Lish et al. (1984), which were slightly different for two groups than those reported in Parker et al. (2006). The resulting BMDs and BMDLs from the remodeling were 64.8 and 32.8 mg/kg-day, respectively, for hepatocellular adenomas and carcinomas and 29.1 and 17.7 mg/kg-day, respectively, for liver and lung tumors combined. See Table D-23 and the subsequent MS-COMBO results for details.

An OSF was derived from the $BMDL_{10\text{-HED}}$ based on a significantly increased trend in the incidence of hepatocellular and alveolar/bronchiolar adenomas or carcinomas in female B6C3F$_1$ mice (i.e., the Liver + Lung $BMDL_{10\text{-HED}}$ from MS-COMBO). The OSF of **$0.04\ (mg/kg\text{-}day)^{-1}$** is calculated by dividing the BMR (10% ER) by the Liver + Lung $BMDL_{10\text{-HED}}$ and represents an upper bound on cancer risk per unit dose associated with a continuous lifetime exposure:

$$OSF = 0.10 \div (\text{Liver + Lung})\ BMDL_{10\text{-HED}} = 0.10 \div 2.66\ mg/kg\text{-}day$$

$$= 3.8 \times 10^{-2}\ (mg/kg\text{-}day)^{-1}$$

$$= 4 \times 10^{-2}\ (mg/kg\text{-}day)^{-1}, \text{ rounded to one significant figure}$$

The slope of the linear extrapolation from the central estimate of exposure associated with 10% extra cancer risk ($BMD_{10\text{-HED}}$) from the same data sets is given by:

Slope of the linear extrapolation from the central estimate

= 0.10 ÷ (Liver + Lung) $BMD_{10\text{-HED}}$ = 0.10 ÷ 4.35 mg/kg-day

= 2.3×10^{-2} $(mg/kg\text{-}day)^{-1}$

= 2×10^{-2} $(mg/kg\text{-}day)^{-1}$ (rounded to one significant figure)

The OSF for RDX should not be used with exposures exceeding the POD (2.66 mg/kg-day), because above this level, the fitted dose-response model better characterizes what is known about the carcinogenicity of RDX.

2.3.4. Uncertainties in the Derivation of the Oral Slope Factor

A number of uncertainties underlie the cancer unit risk for RDX. Table 2-8 summarizes the impact on the assessment of issues such as the use of models and extrapolation approaches particularly those underlying the *Guidelines for Carcinogen Risk Assessment* (U.S. EPA, 2005a), the effect of reasonable alternatives, the approach selected, and its justification.

Table 2-8. Summary of uncertainty in the derivation of the cancer risk value for RDX

Consideration and impact on cancer risk value	Decision	Justification
Selection of study The cancer bioassay in the rat (Levine et al., 1983b) would provide a lower estimate of the OSF	Lish et al. (1984) as principal oral study to derive the human cancer risk estimate	Lish et al. (1984) was a well-conducted study; five dose levels (including control) used, with a sufficient number of animals per dose group (at terminal sacrifice, n = 62–65 female mice/dose group except highest dose where n = 31). Tumor data from the mouse provided a stronger basis for estimating the OSF than rat data. Confidence in the OSF based on rat data was low because of the small numbers of tumors.
Species/sex Use of data sets from the male mouse or male rat would provide a lower OSF	OSF based on tumors in female $B6C3F_1$ mouse	It is assumed that a positive tumor response in animal cancer studies indicates that the agent can have carcinogenic potential in humans in the absence of data indicating that animal tumors are not relevant to humans (U.S. EPA, 2005a). As there are no data to inform whether the response in any given experimental animal species or sex would be most relevant for extrapolating to humans, tumor data from the most sensitive species and sex were selected as the basis for the OSF. Other data sets would provide smaller OSF values, and are not considered any more or less relevant to humans than data from the female mouse (i.e., 0.017 per mg/kg-day based on hepatocellular carcinomas in male F344 rats,

Consideration and impact on cancer risk value	Decision	Justification
		and 0.018 per mg/kg-day based on alveolar/bronchiolor carcinomas in male B6C3F$_1$ mice; see Appendix D, Section D.2).
Combined tumor types Human risk would ↓ if OSF was based on analysis using only a single tumor type	OSF based on liver and lung tumors in female B6C3F$_1$ mouse	Basing the OSF on one tumor site could potentially underestimate the carcinogenic potential of RDX, so an analysis that included data from the two tumor sites was chosen to calculate the combined risk. Because there is no known biological dependence between the liver and lung tumors, independence between the two tumor sites was assumed. This is not likely to produce substantial error in the risk estimates (NRC, 1994).
Selection of dose metric PBPK models are available for the rat, mouse, and human, and using an appropriate internal metric can ↑ accuracy in human extrapolation	Mouse liver and lung tumors: administered dose used	EPA evaluated a published PBPK model in the mouse (Sweeney et al., 2012b); major uncertainties associated with limited toxicokinetic data in the mouse and unknown differences in metabolism across species were identified. Althought EPA's preferred approach for extrapolating results from animal studies to humans is toxicokinetic modeling, the uncertainties associated with use of the mouse PBPK model for RDX were considered higher than use of administered dose.
Cross-species scaling Alternatives could ↓ or ↑ OSF (e.g., 3.5-fold ↓ [scaling by body weight] or ↑ 2-fold [scaling by $BW^{2/3}$])	$BW^{3/4}$ scaling (default approach)	There are no data to support alternatives. Because the dose metric was not an AUC, $BW^{3/4}$ scaling was used to calculate equivalent cumulative exposures for estimating equivalent human risks. While the true human correspondence is unknown, this overall approach is not expected over- or underestimate human equivalent risks.
BMD model uncertainty Alternative models could ↓ or ↑ OSF	Use multistage model to derive a BMD and BMDL for combined tumor incidence	No biologically based models for RDX are available, and there is no a priori basis for selecting a model other than the multistage. The multistage model has biological support and is the model most consistently used in EPA cancer assessments (Gehlhaus et al., 2011).
Low-dose extrapolation approach ↓ cancer risk would be expected with the application of nonlinear extrapolation	Linear extrapolation from the POD	Where the available information is insufficient to establish the MOA for tumors at a given site, linear extrapolation is recommended because this extrapolation approach is generally considered to be health-protective (U.S. EPA, 2005a). Because the MOA for RDX-induced liver and lung tumors has not been established, linear low-dose extrapolation was applied, consistent with EPA guidance.
Statistical uncertainty at the POD ↓ OSF by 1.6-fold if BMD used as the POD rather than the BMDL	BMDL (default approach for	Lower bound is 95% confidence interval (CI) on administered exposure at 10% ER of liver and lung tumors.

Consideration and impact on cancer risk value	Decision	Justification
	calculating plausible upper bound OSF)	
Sensitive subpopulations ↑ OSF to an unknown extent	Considered qualitatively	There is little information on whether some subpopulations may be more or less sensitive to the potential carcinogenicity of RDX (i.e., because of variability in toxicokinetics or toxicodynamics for RDX). The mode of carcinogenic action for liver and lung tumors in experimental animals is unknown, and little information is available on RDX metabolites or variation in metabolic rates that could be used to evaluate human variability in cancer response to RDX.

2.3.5. Previous IRIS Assessment: Oral Slope Factor

The previous cancer assessment for RDX was posted to the IRIS database in 1990. The OSF in the previous cancer assessment was based on the bioassay by Lish et al. (1984) and analysis of data for hepatocellular adenomas or carcinomas in female mice. An OSF of 1.1×10^{-1} $(mg/kg\text{-}day)^{-1}$ was derived using a linearized multistage procedure (extra risk) and scaling by body weight to the 2/3 power for cross-species extrapolation. In addition, the previous assessment dropped the high-dose group because the dose was reduced at week 11 to address high mortality.

2.4. INHALATION UNIT RISK FOR CANCER

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. Quantitative risk estimates may be derived from the application of a low-dose extrapolation procedure. If derived, the inhalation unit risk (IUR) is a plausible upper bound on the estimate of risk per $\mu g/m^3$ air breathed.

An IUR value was not calculated because inhalation carcinogenicity data for RDX are not available. While inhalation absorption of RDX particulates is a plausible route of exposure, there are no toxicokinetic studies of RDX inhalation absorption to support an inhalation model. Therefore, a PBPK model for inhaled RDX was not developed to support route-to-route extrapolation of an IUR from the OSF.

2.5. APPLICATION OF AGE-DEPENDENT ADJUSTMENT FACTORS

As discussed in the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA, 2005b), either default or chemical-specific age-dependent adjustment factors (ADAFs) are recommended to account for early-life exposure to carcinogens that act through a mutagenic MOA. Because no chemical-specific data on lifestage susceptibility for

RDX carcinogenicity are available, and because the MOA for RDX carcinogenicity is not known (see Section 1.2.5), application of ADAFs is not recommended.

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